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T. N. Savinova

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CHEMICAL POLLUTION OF NORTHERN SEAS

by

T.N. Savinova

The monograph presents a review of the ecotoxicological and biogeochemical literature on pollution of northern seas by chlorinated and petroleum hydrocarbons. Data is presented regarding the bioaccumulation of these pollutants in the organs and tissues of seabirds and of hydrobionts at various trophic levels. The effects of pollutants on phytoplankton in northern seas are discussed. Factors affecting the bioaccumulation of toxicants in fish are also examined.

The monograph may be of interest to biologists, toxicologists, public health physicians, environmental specialists, and personnel in the hydrometeorological service. 13 figs., 15 tables, 424 bibliographic entries.

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INTRODUCTION

Contamination of the seas by various toxic substances of anthropogenic origin results in substantial disruptions in the physico-chemical makeup of natural bodies of water and exercises a negative effect on marine organisms and on marine ecosystems as a whole. 5/*

The ecosystems of northern seas, whose pollution levels have received little attention, are particularly sensitive to anthropogenic influences. The seas of Northern European are one of the most important fishing regions in the world, thus lending urgency to biogeochemical and ecotoxicological studies in the area.

Among the organic compounds entering the oceans as a result of man's activity, the greatest attention is focused on petroleum and chlorinated hydrocarbons. These include the organochlorine pesticides (dichlorodiphenyltrichloroethane [DDT] and its metabolites [DDE, DDD), the α and γ -isomers of hexachlorocyclohexane (HCH), kepone, and others) as well as compounds with physico-chemical and chromatographic properties resembling organochlorine pesticides – polychlorinated biphenyls (PCBs). These toxicants are characterized by biological and chemical persistence in marine environments, and by an international convention in 1974 were included in the list of the most dangerous chemical substances, the discharge of which into the ocean was prohibited.

The world's literature contains extensive information about the side effects and long-term effects of biological circulation of chlorinated and petroleum hydrocarbons in freshwater and marine ecosystems. In the 60' and 80's there appeared ground-breaking studies describing the above-mentioned toxicants as specific ecological factors in the life of marine bodies of water. These works included monographs by R. Carson (1967), L.P. Braginskii (1972), L.P. Braginskii et al. (1979, 1987), S.A. Patin (1979), S. Gerlach (1981), and O.G. Mironov (1971, 1985). The principles of ecological and climatic monitoring were set out in the monograph of Yu.A. Izrael (1984).

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* Figures in the right-hand margin indicate page numbers of the original (Tr.).

The present monograph reviews the literature on pollution levels in northern seas due to petroleum and chlorinated hydrocarbons and the results of the 6/ author's biogeochemical and ecotoxicological research carried out at the Murmansk Marine Biology Institute (MMBI) of the USSR Academy of Sciences' Kola Scientific Centre. The author wishes to thank the associates of MMBI's laboratory of hydrology and hydrochemistry and plankton laboratory for their assistance in the conduct of field and experimental work, and the author is deeply grateful to candidates of chemical sciences V.I. Kofanov and S.M. Chernyak for their assistance with chemical analysis.

In an effort to reflect the great complexity involved in fully clarifying such a multifaceted problem as pollution of northern seas and their biological resources, the monograph pays particular attention to reviewing the disparate and rather limited information regarding levels of toxic substances in commercial marine organisms and to analyzing the effects of pollutants on natural communities of phytoplankton in northern seas and on seabird populations. The author hopes the monograph will be of interest to hydrobiologists, toxicologists, to specialists in environmental protection, and to all those concerned about solving what is the most important problem today – protecting the ocean's biological resources.

Chapter 1. CHLORINATED AND PETROLEUM HYDROCARBONS: PRODUCTION AND USE

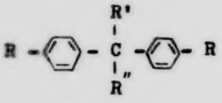
Chlorinated hydrocarbons include organochlorine pesticides (DDT and its metabolites, HCH, kepone, and others) and polychlorinated biphenyls (PCBs).

The term DDT (dichlorodiphenyltrichloroethane) is in use throughout the world and stands for 1,1' - (2,2,2-trichloroethylidene)-bis chlorobenzene (p,p'-DDT). The chemical structure of certain DDT analogs (p,p'-isomers) is given in Table 1. The structure of o,p'-isomers can be derived from the structure of p,p'-isomers.

DDT was first synthesized in Strasbourg in 1873 by S. Zeidler. In 1939 the Swiss scientist P. Müller discovered the insecticidal properties of DDT. In 1940, when wide use of this substance began, it was considered the ideal weapon for use in man's battle with insect pests that damage crops and carry disease. After large yields of grain were achieved everywhere with the aid of DDT, Müller was awarded a Nobel prize in 1948 (Randers, 1973). DDT was used with success to combat more than 3000 species of agricultural pests. Thanks to DDT, by 1955 malaria had been eradicated in 37 countries with a population of 728 million people (Metcalf, 1973). According to Edwards (1974), during its first 10 years of use DDT saved over 5 million human lives and brought in agricultural profits in the USA estimated at 21 billion dollars. But the first scientists' voices against use of DDT were raised as early as 1946 (Cottam, Higgins, 1946) when it was discovered how sensitive fish and crustaceans are to DDT, and it was concluded that this substance should never be used near bodies of water.

Table 1[1] 8/

Structure of p,p'-DDT and its analogs in the form



Compound	Chemical name	R	R'	R"
DDT	1,1'--(2,2,2-trichloroethylidene)-bis-4-chlorobenzene	-Cl	-H	$-CCl_3$
DDE	1,1'-(2,2-dichlorovinylidene)-bis-4-chlorobenzene	-Cl	none	$=CCl_2$
DDT TDE	1,1'-(2,2-dichloroethylidene)-bis-4-chlorobenzene	-Cl	-H	$-CHCL_2$
DDA	2,2-bis-(4-chlorophenyl)-acetyl acid	-Cl	-H	-C(O)OH

The late 50's and 60's saw intensive study of the properties of DDT and its effect on the environment. It was revealed that bioaccumulation is one of the most characteristic features of organochlorine pesticides, due to their physico-chemical properties. All of the very factors that make DDT such a convenient insecticide (low vapor tension, low solubility in water and high solubility in fats, and a general resistance to biological oxidation and photo-oxidation) contributed to making DDT the ideal prototype of an environmental pollutant.

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[1] Tables and figures occur in the translation in the same order as in the original. Where practical, tables and figures also appear at the same point in the text as in the original. Otherwise, they appear soon after the first point in the text at which they are cited. (Tr.)

Although there are no precise data concerning world DDT production, specialists estimate that from 1940 to 1970 more than 2 million tonnes of DDT were used throughout the world (Woodwell et al., 1971; Metcalf, 1973). Table 2 presents some figures on world consumption of DDT.

Table 2 9/

Annual pesticide use in various countries

Country	Year	Quantity, kg/ha	Source
USA	1970	1.8	Novozhilov, 1975
USSR	1970	1.31	"
Japan	1970	11.4	"
FRG	1970	12.0	"
USA	1976	2.92	Production Yearbook, 1977
India	1976	0.265	"
Japan	1976	16.6	"
Africa	1970	2.7	Braginskii, 1972
India	1983-1984	0.855	Cherkasova, 1978

Following the appearance of R. Carson's book *Silent Spring* (1962) there was a tendency to treat pesticides more carefully. The Environmental Protection Agency (EPA) was created in the USA, which has the authority to restrict or prohibit use of dangerous chemical substances. In the 70's the use of DDT was prohibited in the USSR (*Spravochnik po pestitsidam* [Pesticide

Handbook], 1974). Use of DDT has also been prohibited in the majority of socialist countries as well as in Belgium, Greece, Ireland, Cyprus, Luxembourg, Netherlands, USA and Finland, and use of DDT is restricted in England, Denmark, Israel, Spain, Norway, Turkey, France and the Federal Republic of Germany (Plant ..., 1975). DDT use, even in limited quantities, is dictated by economic necessity, but there are already a number of situations in which this same necessity forces its abandonment. The USA, for example, has refused to purchase meat from Argentina because of its high levels of DDT (Mani, 1970). The Americans have stopped using swordfish in foods for the same reason, but are exporting it to countries with less stringent legislation, just as the Danes are exporting cod liver to the countries of Southern Europe (Chevallier, 1978). Developed capitalist countries are exporting significant quantities of organochlorine pesticides to developing countries. The latter are currently experiencing a critical phase in the use of pesticides since, at current rates of use, the very same problems may appear as occurred in the developed countries, and to an even greater extent.

HCH (hexachlorocyclohexane, hexachlorane, 'hexatox'[2], 'pultax', 'sinex', 10/
'yakutan') is a combination of 8 isomers with the empirical formula $C_6H_6Cl_6$ and a relative molecular weight of 290.8.

The most active insecticidal properties are possessed by the γ-isomer ('agrizert', 'VNS', 'hematox', lindane, 'lindaram', 'lindatox'); the technical form of HCH contains no less than 90% γ-isomer of HCH. Lindane is widely used to combat plant pests throughout the world. Some idea of the extent of HCH production and use can be gained from the fact that in 1968, 45 695 tonnes of technical HCH were produced in Japan alone (Edwards, 1974).

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[2] The rendering of some names could not be verified. In the translation these appear in single quotation marks. (Tr.)

Kepone ('chlorodecone') – 'decachlorotetracyclodecanon' – is a colourless crystal with the empirical formula $C_{10}Cl_{10}O$ and relative molecular weight of 460.6. Kepone dissolves poorly in water and well in acetone, benzene and alkalies. It is manufactured in the form of a 50% wettable powder, a 50 and 10% dust and 10% granules. It is used to combat agricultural pests as well as the larvae of flies and termites.

Polychlorinated biphenyls (PCBs) have been used in industry since 1929. Their dielectric properties permit their use both as coolants and as insulating substances in sealed heating and electrical systems. In the USA up to 60% of the PCBs produced are used for these purposes. Approximately 25% are used as plasticizers of technical polymer materials. Due to their thermal and fire resistance, up to 10% of PCBs are utilized as high-pressure hydraulic fluids, heat transfer agents, and specialized lubricants for cutting devices. PCBs are employed in the production of protective coatings, adhesive agents, printer's ink and duplicating paper, as pesticide fillers (lindane, chlordane, aldrin, dieldrin, toxaphene, and others) to reduce the volatility of substances, and in some cases to improve insecticidal properties (Lichtenstein et al., 1969).

Wide use of PCBs has contributed to environmental pollution. The toxicity of PCBs is rather high, with some known cases of fatal poisoning of humans (Finklea et al., 1972).

Jensen (1966) was the first to identify PCBs in specimens of fish and birds. 11/
Subsequent studies indicated a wide distribution of these substances in the biosphere (Jensen et al., 1969); Risebrough et al., 1968a, 1972; Johnston,

1976; Williams, 1979; Falandysz, Szefer, 1984; Mearns, 1986; Cosper et al., 1988).

Despite this, however, world-wide production of PCBs remains rather high – 100 million t per year. PCBs were most widely used in the 60's and 70's. In the USA PCB use is currently approximately 65 000 t annually (McCraw, 1983), with over 4500 t entering the environment each year (Maugh, 1975). PCBs are manufactured in various countries under the following brand names: Aroclor (Monsanto Company, USA); Phenoclor, Pyralene (Prodelec, France); Kannechlor, Santotherm (Kanegafuchi Chem. Co. & Mistibushi [sic (Tr.)]-Monsanto, Japan); Clophen (I.G. Farben-Industrie A.G., Germany); Fenclor (Caffaro, Italy) [Karim, 1976]; Sovol (USSR) [Kryzhanovskaya, Shirokaya, 1975].

In recent years measures have been undertaken in West European countries to restrict the PCB use, and the rate of their production has decreased. However, although the use of PCBs has been prohibited since 1971 in Sweden, since 1973 in Denmark and (partially) since 1973 in Finland (Kihlström, Berglund, 1978), the residual concentrations of these substances in living organisms remain high and represent a serious danger for terrestrial and aquatic ecosystems.

Prior to the beginning of the 20th Century, petroleum products were used primarily for lighting. By the beginning of the century liquid fuel engines were coming into wide use, and with the development of electrical power petroleum products began to be used principally as a fuel. Today oil and petroleum products account for 50% of the world's energy requirements, including one-third of electrical power generation. Over the past 50 years oil demand has doubled every 10 years, and these rates are expected to continue in the future

(Jagger, 1971). Military requirements in the years 1939-1945 along with reductions in the mining of coal contributed to development of the petrochemical industry. In 1938 the world oil output stood at approximately 278 million t, of which Western Europe consumed 36 million t. By the mid-50's these figures had grown 4-fold, and have increased 10-fold today (Nelson-Smit, 12/ 1977).

Today the Middle East is the most important oil-producing region, although recent studies in Alaska indicate significant reserves of oil, and oil reserves in the North Sea could soon satisfy most of the requirements of Northern Europe.

In Geneva in 1958 an international convention was signed that proclaimed coastal countries' sovereignty over the continental shelf adjacent to their borders, giving them the exclusive right to the exploration and development of mineral resources. In the late 50's industrial reserves of natural gas were discovered on the southeast coast of the North Sea near the Dutch city of Groningen, and preparations began for prospecting on the shelf. In the early 60's oil prospecting was begun on the North Sea shelf. The countries bordering the North Sea divided it into sectors. As a result, most of the continental shelf ended up in the possession of England and Norway. In 1965 the Norwegian part of the shelf south of the 62nd parallel was broken up into 278 sections. In May 1970 industrial quantities of high-quality oil were discovered in the "Ekofisk" area, and by the end of the year oil had been found in ten other areas. In 1976 Norway became the first country to satisfy all of its petroleum requirements from oil recovered on the continental shelf.

By the mid 70's the contours of a massive deposit had been drawn in the central portion of the North Sea. Experts estimates that the deposit, known as the 'Statfjord', contains 14 billion t of oil and 100 billion cubic metres of gas.

By decision of the Norwegian Storting, exploratory drilling north of the 62nd parallel began in the 80's. In 1983 ten floating drilling platforms were in operation on the Norwegian shelf. In recent years production has dropped off somewhat at the largest "Ekofisk" oil and gas deposit, yielding 12 to 15 million t of oil annually. (In 1980 this deposit accounted for 75% of total production.) Construction of an oil pipeline is planned from the new oil and gas deposit 'Gulfaks' to Bergen (a distance of approximately 200 km), and the pipeline will be extended over land to the oil refinery in Mengstad. Since 1980 oil and gas production in the Norwegian sector of the North Sea has remained stable at a level of approximately 50 million t of petroleum equivalent (Rodionov, 1985). 13/

According to Gerlach (1981) exploratory drilling on the shelfs of the world's oceans had been conducted at 20 000 sites in the preceding 30 years. Over 1000 sites have been counted in the North Sea alone, 350 of which are in operation (Fig 1). All together, as of July 1, 1980, 782 sites had been developed on ocean shelfs, and these had yielded 9.3 billion t of oil and approximately 3.3 billion cubic metres of gas (Levchenko, 1982). In the last few years active exploratory drilling has been underway on the shelf of the Barents Sea. 14/

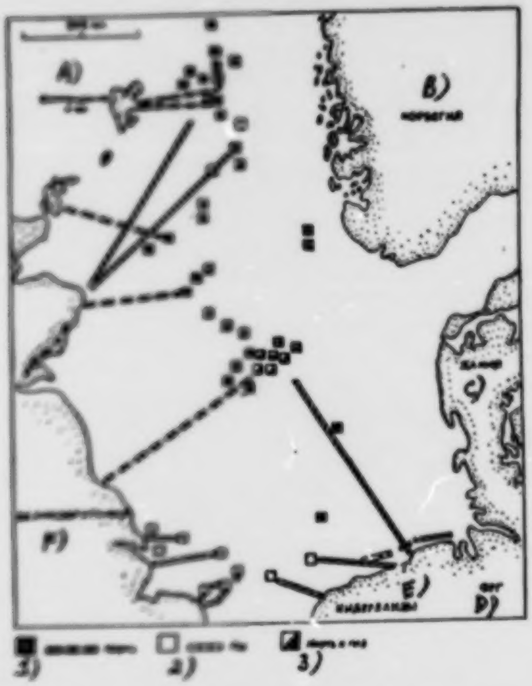

Рис.1. Разведка нефти, нефтяные и газовые трубопроводы в Северном море (Gerlach, 1981).

■– – – 1) – нефть; □—— 2) – газ; ◪ 3) – нефть и газ.

Figure 1. North Sea oil exploration and oil and gas pipelines (Gerlach, 1981).
[A- Shetland Islands; B- Norway; C- Denmark; D- FRG; E- Netherlands; F- Great Britain.
1- oil; 2- gas; 3- oil and gas.

Crude oil and its derivatives are an exceptionally complex mixture of numerous chemical compounds. The basic components of oil are hydrocarbons (up to 98% of the total mass) and their derivatives, containing oxygen, sulphur and nitrogen. The hydrocarbons in oil can be divided into four classes:

1. Paraffins - stable[3] saturated compounds described by the formula 'C_nH_{2n+2}' and having a straight branching chain. The number of carbon atoms in the paraffins found in oil varies from 6 to 17; their boiling point varies from 5.5 to 22° C, density from 0.66 to 0.78 g/cm^3, solubility in water from 3 to 138 μg/l, and they usually make up 19-25% of the crude oil.

2. Naphthenes (cycloparaffins)- saturated cyclic compounds described by the formula 'C_nH_{2n}', both hydrogen atoms of which can be replaced by alkyl groups. Naphthenes in crude oil contain 5 to 9 carbon atoms, have a boiling point from 49.3 to 141.2° C, and a density of 0.75-0.79 g/cm^3. Their water solubility is negligible. They usually account for 4-10% of the crude oil.

3. Aromatic unsaturated cyclic compounds of the benzene series having a ring with six hydrogen atoms fewer than the corresponding naphthene ring. Their hydrogen atoms can also be replaced by alkyl groups. The number of carbon atoms in the aromatic compounds of crude oil varies from 6 to 13, the boiling point from 80 to 354° C, density from 0.87 to 1.25 g/cm^3, solubility in water from 20 to 840 μg/l, and they account for 2-40% of the crude oil.

4. Olefins (alkenes) – unsaturated non-cyclic compounds. These are the principal product of petroleum cracking and a product of their [sic (Tr.)] oxidation.

The ratios of various classes of hydrocarbons in oil are of great significance with regards to use and biological [effects] inasmuch as the aromatic compounds are more toxic than the paraffins.

The key physical characteristics of oil – boiling point, specific weight (density) and viscosity – are determined by the chemical nature and ratio of its 15/

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[3] Russian 'ustoichivyi' can also be translated as 'persistent' and is sometimes so translated, depending on the context (Tr.).

components. The boiling point of oil increases with an increase in molecular weight, density depends primarily on molecular structure, and viscosity depends on both factors. The higher [molecular weight] paraffins determine the solidifying point and viscosity of oil.

Chapter 2. DISTRIBUTION OF CHLORINATED AND PETROLEUM HYDROCARBONS IN A MARINE ENVIRONMENT. THEIR TRANSFORMATION AND BIODEGRADATION

2.1 Chlorinated hydrocarbons

A significant number of studies have been devoted to the distribution of persistent chlorinated hydrocarbons in the biosphere and the pathways by which they enter water (Risebrough et al., 1968a, b; Risebrough, 1969; Seba, Prospero, 1971; Addison, 1976; Bidleman et al., 1981; Gerlach, 1981; Tanabe et al., 1982; Braginskii et al., 1979; Patin, 1979; Izrael, 1984, etc.).

The well-known American ecologist Woodwell (1967) advanced the suggestion that the findings obtained from data on the migration and precipitation of radioactive fallout are also applicable to chlorinated hydrocarbons. Based on an analysis of numerous studies, Woodwell came to the conclusion that DDT is distributed in the biosphere through the following basic mechanisms: 1) by distribution through air currents, which the DDT enters in the course of aerial dusting of forests and farm crops; 2) by means of ocean surface currents; 3) by biological mechanisms of toxic substance concentration. In addition, in coastal regions direct runoff from agricultural land and waste water discharge in the vicinity of large industrial centres are quite significant 16/ (Butler et al., 1970; Patin, 1977, 1979).

Between 10 and 70% of pesticides are known to be released in the atmosphere during use (Hurting, 1972). Approximately 50% of DDT enters the atmosphere through evaporation from upper soil layers (Lloyd-Jones, 1971).

The mechanisms by which various substances form gaseous and aerosol pollutants and these are transported to the sea surface differ substantially, but in all cases this leads to global contamination of the world's oceans. The contribution of atmospheric origin to the total input of pollutants into the marine environment amounts to 80-90% (Patin, 1977). American scientists made a major contribution to studying this issue following completion of the 2-year Program to Study the Transport of Pollutants[4], carried out in the framework of the International Decade of Ocean Exploration (Duce et al., 1972, 1974; Risebrough, 1972; Harvey, Steinhauer, 1974b; Bidleman, Olney, 1974). Research in this area was later continued (Holden, 1976; Harvey, Steinhauer, 1976; Bidleman et al., 1981; Tanabe et al., 1982). Soviet scientists devoted studies to clarifying the role of Atlantic currents in the transport of polluting substances (Simonov et al., 1974; Mikhailov, 1985; Orlova, 1985; Simonov, 1985; Savinova et al., 1982).

One of the first attempts at constructing a dynamic model of DDT in an ecosystem was that of Harrison et al. (1970). Subsequently, when it was understood that the biota is not the principal accumulator of DDT, more accurate models for the circulation of this substance appeared (Woodwell et al., 1971; Cox, 1972; Cramer, 1973). Figure 2 shows a schematic which more fully reflects the migration pathways of toxicants in aquatic ecosystems (Braginskii et al., 1987).

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[4] The official name of this program could not be verified (Tr.).

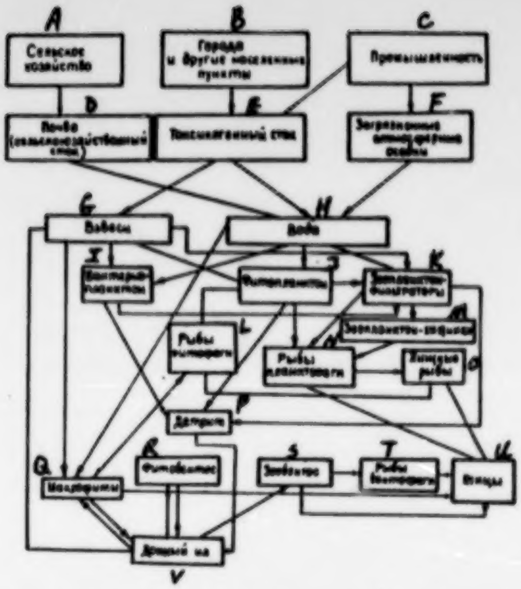

Рис.2. Миграционные пути токсикантов в водных экосистемах (Брагинский и др., 1987).

Figure 2. Toxicant migration pathways in aquatic ecosystems (Braginskii et al., 1987). [A - Agriculture; B - Cities and other populated sites; C - Industry; D - Soil (agricultural runoff); E - Toxicogenic runoff; F - Contaminated atmospheric precipitation; G - Suspended matter; H - Water; I - Bacterioplankton; J - Phytoplankton; K - Zooplankton filtrators; L - Phytophagous fish; M - Predator zooplankton[5]; N - Planktophagous fish; O - Predator fish; P - Detritus; Q - Macrophytes; R - Phytobenthos; S - Zoobenthos; T - Bottom-feeding fish; U - Birds; V - Bottom silt.]

In the opinion of American experts (NAS, 1971) the half-life of chlorinated hydrocarbons in marine ecosystems is measured in years, decades and, possibly, in centuries. The process of DDT degradation follows an exponential curve, making it possible to talk about a half-life which, according to various authors, ranges from 7-17 to 38 years (Kaplin, 1967; *Spravochnik po* 17/ *pestitsidam,* 1974; Bloom, Menzel, 1971; Armstrong, Weimer, 1973).

Of major importance for the transformation of organochlorine pesticides are the processes occurring at the interface between various media: evaporation, adsorption, desorption from decay products, penetration into bottom deposits, and anaerobic metabolism. The adsorption coefficient for DDT reaches 10^3 (Harvey et al., 1974).

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[5] Russian 'zooplankton-khishchniki', literally 'zooplankton-predators'. Another possible rendering might be 'zooplankton predators' (Tr.).

Chlorinated hydrocarbons may degrade under the influence of sunlight. Long-wave and ultraviolet solar radiation plays a substantial role in this process of photooxidation of certain pesticides and their metabolites (290-400 μm) (Miller, Narange, 1970; Parmar et al., 1976; Zeep et al., 1977).

Experimental data (Maugh, 1973) indicates that DDT is capable of being converted into PCB isomers under the impact of UV radiation, and this may be 18/ one of the sources of the growing PCB contamination of aquatic environments observed recently.

In aerobic conditions and in the presence of certain microorganisms DDT is converted in bodies of water into DDE, while in anaerobic conditions decay proceeds by formation of DDD (Chacko et al., 1966; Wedemeyer, 1967; Guenzi, Beard, 1967; Ware, Clifford, 1970; Whitacre et al., 1972).

General descriptions for the basic transformation pathways of DDT have been presented by Metcalf et al. (1971) and Khan et al. (1976), the latter of which is shown in Figure 3.

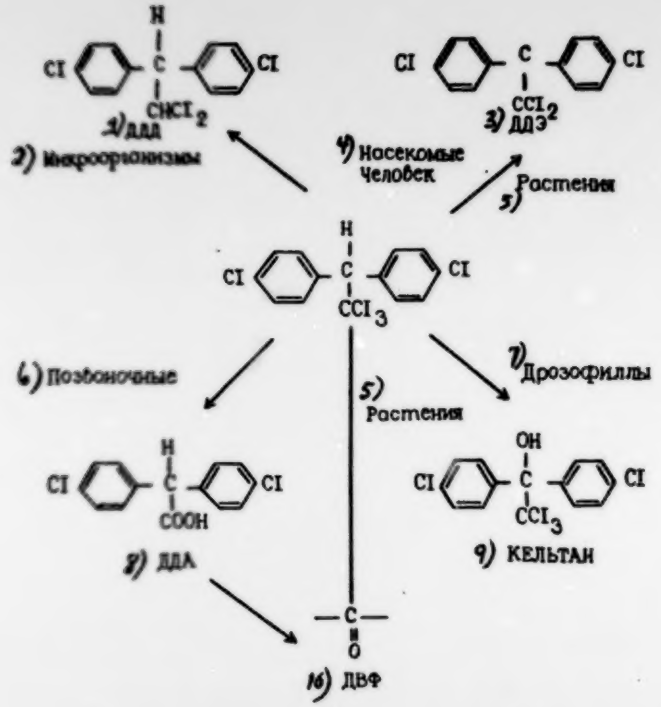
Cl
H
C
CHCl2
1) ДДД
2) Микроорганизмы
Cl
C
CCl2
3) ДДЭ
4) Насекомые Человек
5) Растения
H
C
CCl3
6) Позвоночные
5) Растения
7) Дрозофиллы
H
C
COOH
8) ДДА
OH
C
CCl3
9) КЕЛЬТАН
C
O
10) ДВФ

Рис.3. Основные пути трансформации ДДТ (Khan et al., 1976).

Figure 3. Primary pathways of DDT transformation (Khan et al., 1976).
[1) DDD; 2) Microorganisms; 3) DDE; 4) Insects/Man; 5) Plants; 6) Vertebrates; 7) Drosophyliae; 8) DDA; 9) Kelthane; 10) Dimethyldichlorovinylphosphate]

Diatoms play a definite role in DDT degradation in the ocean (Rice, Sikka, 19/ 1973). Marine invertebrates are also capable of breaking down DDT into DDE and DDD (Brown, 1971). It has also been shown that DDT can be broken down by microflora in the intestine of fish: rainbow trout (Wedemeyer, 1966, 1968; Addison, Zinck, 1977), anchovies (Malone, 1970), Atlantic salmon (Greer, Paim, 1968; Cherrington et al., 1969; Addison et al., 1976).

Degradation of PCBs by microbial cells depends on the structural characteristics of these compounds and the number of chlorine atoms in them. Mono- and dichlorobiphenyls, as well as compounds with an unsubstituted ring, degrade comparatively rapidly from the action of microorganisms. Biphenyls containing more than four chlorine atoms are extremely resistant to biodegradation (Metcalf et al., 1975; Sayler et al., 1978; Kalmaz, 1979; Lui, 1982).

2.2 Petroleum hydrocarbons

Various estimates exist for the quantity of petroleum hydrocarbon (PH) input into the marine environment – from 3 to 28 $\cdot$ 10^6 t/yr (Barbier et al., 1973; Goldberg, 1976). The most complete evaluation was carried out by the National Academy of Sciences of the USA (NAS, 1975). According to this evaluation, approximately 6 $\cdot$ 10^6 t PHs are currently entering seas and oceans from various sources and through various pathways, which amounts to approximately 0.23% of annual global oil production. The primary sources and pathways by which PHs enter the world's oceans can be pictured as follows (%): discharges from ships at sea of wash and ballast water -- 23: discharges from ships in port and losses during offloading of oil[6] from tankers – 17; industrial and domestic effluents – 11; municipal rainwater runoff – 5; spillage from ship disasters – 5; from drilling on continental shelfs – 1; from rivers – 28; from atmospheric sources – 10.

The initial investigative stage in research into the contamination of seawater by petroleum hydrocarbons had been completed by the mid 70's. As a result of this research a number of general conclusions were made about the nature of marine pollution. The most important conclusion was that PH contamination 20/

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[6] The Russian here uses 'toplivo' - 'fuel' - rather than the expected 'neft' - 'oil' (Tr.).

was becoming a global phenomenon. At the present time PHs are just as likely to be encountered in mediterranean type seas, in open seas and even in the open ocean. Pollution zones form along the coasts and then spread far beyond coastal areas, affecting numerous entire seas and wide expanses of ocean; these zones are stable both in time and in space (Simonov, 1978). Global zones of marine pollution are characterized by maximum levels in the euphotic layer, by elevated contamination of ecosystems of the neritic zone and inland seas, by zonality in the distribution of toxicants and the latitudinal effect, by the "mosaicness" of toxicant concentrations in the water, by their localization in the biotope of the hyponeuston and benthos, and by the superimposition of zones of maximum pollution in areas of high biomass and productivity (Patin, 1977).

Upon entering the sea, oil usually forms slicks – a surface film of variable thickness. During the first few hours in the existence of a slick physico-chemical methods predominate in removing PHs from the water's surface. Components with a low boiling point quickly evaporate, carrying with them the most highly volatile fractions (Berridge et al., 1968), with the result that a significant quantity of compounds containing up to eight carbon atoms are carried into the atmosphere. In the first few days, depending on the composition of the oil and on hydrometeorological conditions, 30-70% of the oil is lost (principally the C_4-C_{12} fractions).

All the components of oil are soluble in water to one degree or another and capable of rapid leaching as the oil spreads on the surface of the water and forms slicks. The high molecular [weight] constituents of oil, which do not dissolve well in water, form various types of emulsions which are very stable and which contain up to 70-80% water. It has been suggested that the cause of their stability lies in the influence of the products of microbial hydrocarbon metabolism. It has been shown that yeasts (*Candids*) and certain bacteria (*Pseudomonas, Nocarida, Mycobacterium*) release lipid-like compounds into the environment which emulsify hydrocarbons and crude oil in water (Khailov, 1971; Tsyban, Simonov, 1978).

The kinetics of oil degradation under static test conditions in the laboratory follow the law of a first-order monomolecular reaction and imitate free-radical 21/ processes (Simonov, 1974). As early as 1950 Frank (1950) demonstrated that auto-oxidation of hydrocarbons in liquid phase presupposes the existence of two successive reactions: a free radical chain reaction ending in formation of hydrogen peroxide, and its degradation with formation of free radicals. The products formed in the degradation of the peroxide may serve as the initiators of subsequent oxidative processes. Auto-oxidation is suppressed by hydrocarbon proteins and phenols. Numerous sulfur-containing compounds in oil also inhibit oxidation, while heavy metals found in oil, such as vanadium, serve as catalysts of oxidation processes.

Paraffins and aromatic hydrocarbons undergo atmospheric oxidation or photooxidation in sunlight, due primarily to the ultraviolet portion of the spectrum (Nesterova, Simonov, 1979). It has been calculated that oxidation of a surface slick in sunlight may total 2 t of oil per day per km^2 of ocean surface (Tsyban, 1970), while lower estimates (Goldberg, 1976) suggest that a 100-tonne slick 8 km^2 in area with an average[7] thickness of 0.02 mm should lose approximately one tonne of oil per day. Experimental studies of sea water from the Kolskii Zaliv [Gulf of Kola], Barents Sea (with its characteristic microbiocoenoses) have shown that shielding samples of water from the effects of ultraviolet radiation lowers the rate of oil degradation almost two-fold (Sokhina, Shcherbakov, 1984).

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[7] The Russian 'srednii' is variously translated as 'average' or 'mean'. In the present work it has been translated 'average' throughout. (Tr.)

It has been shown experimentally that temperature is a determining factor in the kinetics of oil and petroleum product breakdown. In general, the rate of a chemical reaction as temperature rises by 10°C increases by a factor of 2-4, while a drop in the temperature of the environment[8] substantially slows both physico-chemical and also biochemical processes linked to degradation and transformation of various substances since temperature conditions exercise an effect on the rates of reproduction of bacterial mass, and a decrease in temperature reduces the overall population and physiological activity of microorganisms (Tsyban, 1970).

An increase in the salinity of sea water also has a negative effect on biochemical oxidation of PHs. A 1% change in salinity alters the half-life of PHs 22/
by 22 hours. But for any marine region changes in salinity are generally quite insignificant, with sharp salinity gradients observed principally in zones affected by river runoff and the melting of ice. For the ocean, changes in salinity are even less significant than for individual seas. The same can also be said for the influence of environmental pH[9] on biochemical oxidation of PHs. Taking into account the maximum range of pH change in a marine environment, the change in PH half-life is within a range of 48 hours. The presence of various organic substances such as phenols, SPAV[10], etc. noticeably retards – and on occasion completely halts – biochemical oxidation if the concentration of these

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[8] Or 'medium' (Tr.).

[9] The Russian 'pH sredy' could also be understood as 'pH of the medium'. The term 'sreda' is variously translated as 'environment' or 'medium'. (Tr.)

[10] Expansion unavailable (Tr.).

substances is an order of magnitude higher than the concentration of PHs (Zatuchnaya, Bakum, 1978).

At the present time the ocean's of the world are known to contain over 100 species of bacteria capable of breaking down petroleum products (Mironov, 1971). Tests of 45 strains of microbial cultures extracted from waters of the Barents Sea and capable of utilizing petroleum products have shown that the more active strains of microorganisms can degrade up to 90% of oil input following a 20-day incubation at room temperature (Izgoreva, Nalbandov, 1975). Experimental data indicates that microorganisms of northern seas are capable of an intensive degradation of petroleum products at temperatures close to 0-2°C (Gusev et al., 1980; Sokhina, Shcherbakov, 1984), which confirms the hypothesis about the substantial role of psychrophilic bacteria in the degradation of PHs at low temperatures (Byrom, Beastall, 1977). But studies conducted in the vicinity of Severnaya Zemlya at various times of year have shown that evaporation plays a key role in the processes whereby Arctic waters clean themselves of oil pollution, that the population of hydrocarbon-oxidating microorganisms was extremely low, and that biodegradation of the oil played no substantial role (Ilinskii et al., 1986).

Hydrocarbon metabolism by marine organisms (with the exception of bacteria) has received little attention. Some data exists on the capability of marine algae to transform PHs (Blumer, 1971). It has been established that zooplankton absorb oil and subsequently eliminate it in the form of fecal matter (Conover, 1971), and that significant quantities of oil are found in the intestine and fecal matter of copepods and barnacle larvae. Estimates indicate that a 23/ single *Calanus finmarchicus* may ingest up to $1.5 \cdot 10^{-4}$ g of oil per day. Thus, a population with a density of 2000 specimens/m^3 occupying an area of one km^2

and 10 m deep can remove 3 t of oil per day from a body of water (Parker, 1985). In recent years works have appeared which contain data on the detoxification of petroleum hydrocarbons by mollusk microsomal enzymes (Livingstone et al., 1986; Moore et al., 1987). English scientists have proposed using detoxification reactions for early diagnosis in the monitoring of oil pollution.

Chapter 3. LEVELS OF CHLORINATED AND PETROLEUM HYDROCARBONS IN THE WATERS OF NORTHERN SEAS

3.1 Chlorinated hydrocarbons

Judging from the numerous studies, it can today be considered a generally accepted fact that such chlorinated hydrocarbons as DDT and its metabolites, α- and γ-HCHs and PCBs are present everywhere in sea water. The presence of residual chlorinated hydrocarbons in Antarctic and Arctic ecosystems is one of the most obvious proofs for the global distribution of persistent pollutants (Tatton, Ruzicka, 1967; Risebrough et al., 1976a,b; Yanchinski, 1980; Tanabe et al., 1982; Lukowski, Ligowski, 1987, etc.).

Although there are exists a good deal of literature making it possible to evaluate the effect of chemical contaminants on marine organisms, there has been little attention paid to the geography of contaminant distribution in the oceans and its variability with respect to time. Without this data, however, it is impossible to formulate scientific recommendations on measures to protect waters against pollution.

S.A. Patin (1979) characterized the structure and dynamics of global technogenic substance distribution in the world's oceans as having the following basic traits: maximum pollution of the euphotic layer, elevated contamination of the neritic zone and of inland seas, zonality of distribution and the latitudinal effect, the "mosaicness" of toxicant distribution and their localization in a surface film, superimposition of maximum pollution levels and high bioproductivity, and the relative stability of contaminant flows and levels. 24/

The total DDT level (ΣDDT) in surface waters of the North Atlantic near the coast of Canada in the late 1960's reached 500 ng/l (Yule, Tomlin, 1970). Maximum concentrations were observed, as a rule, in estuaries and coastal regions (Lovelock et al., 1973; Cole, 1973, 1974), thereby creating a threat to commercial and sport fishing. In the USA it has been officially acknowledged (NAS, 1971) that DDT is the seventh most toxic – and DDD the twelfth – among the particularly dangerous organochlorine compounds, the level of which should not exceed 50 ng/l in bodies of water. In the early and mid 70's, however, the ΣDDT concentration in surface waters of the North Atlantic did not exceed 1 ng/l (Harvey et al., 1973; Orlova, 1977, 197[8?]; Brugmann, Luckas, 1979). Jones and Phaender (1976) studied the distribution of organochlorine pesticides (OCPs) in waters of the North Atlantic (from the surface to a depth of 1000 m) and determined the average concentration in residual quantities of DDE at 3.8 ng/l. In the surface layer, however, the average ΣDDT concentration did not exceed 4 ng/l. The levels of organochlorine pesticides for a 4-year period of observation by researchers of GOIN [State Oceanographic Institute (Tr.)] are presented in Table 3.

Table 3

Change in the concentrations of organochlorine pesticides in the surface layer of the North Atlantic for the period 1977-1980, ng/l (Orlova, 1985)

OCP	1977	1978	1979	1980
DDT	19.2-0(1.2)	42.4-0(0.8)	2410-0(0.9)	17.6-0(0.8)
DDE+DDD	3.8-0(0.6)	3.6-0(0.2)	6.1-0(0.6)	1.8-0(0.3)
HCH	2.8-0(0.5)	5.2-0(0.3)	4.2-0(0.2)	4.9-0(0.1)

NOTE: Average concentration given in parentheses.

The lowest level of DDT and its metabolites for the entire period of research 25/ was found in the sub-Arctic zone: average ΣDDT level in this region in 1977 was 0.4 ng/l, 0.2 in 1978, 0.4 in 1979 and 0.2 in 1980. The situation with HCH was different, with maximum concentrations found precisely in the sub-Arctic zone (Orlova, 1985).

In the North Atlantic the vertical distribution of residual organochlorine compound concentrations attains depths of 500 m and greater (Jones, Phaender, 1976); Orlova, 1978). The level of concentrations in lower layers is of approximately the same order as that in the surface layer of the water.

Chlorinated hydrocarbons enter the ocean primarily from the atmosphere, so that these pollutants accumulate at the interface of these two media. GOIN scientists are engaged in studying the levels of accumulation of organochlorine pesticides in the surface microlayer (SML) of the North Atlantic (Mikhailov, 1978; Simonov, Mikhailov, 1979; Mikhailov et al., 1982). It has been shown that the concentration of OCPs in the SML in the northeastern part of the Atlantic Ocean reaches 100 ng/l. Their distribution in the water is irregular, with significant concentrations of OCPs found particularly in the SML in shelf zones of Ireland (more than 100 ng/l), Iceland (approximately 80 ng/l) and Norway (up to 80 ng/l) as a result of coastal runoff.

It should be pointed out that, in these areas, DDT metabolites account for more than 60% of the DDT total. The area through which the North Atlantic current passes is characterized by a change in concentrations of OCPs in the SML from 10 to 30 ng/l. As this water mass moves eastward the concentrations of OCPs decrease to 20 ng/l, which is linked to a significant transformation in this water mass, to dispersal of these substances in space, and to evaporation. An adequacy of DDT concentrations[11] and its metabolite totals have been recorded in the area where the North Atlantic current passes. The HCH concentrations in the SML in this region range from 0 to 70 ng/l. A distinctive feature of the HCH distribution in the SML is its insignificant quantities (less than 10 ng/l) in shelf waters of Ireland and Great Britain.

Elevated concentrations of OCPs in the SML in certain regions of the Northeast Atlantic are confined to areas of oil and gas production on the shelf 26/ since OCPs entering the sea from the atmosphere or from coastal runoff dissolve in the petroleum hydrocarbons concentrated here. V.I. Mikhailov (1985) analyzed the relationship between PH concentrations and total OCPs in the SML of these regions. The results obtained suggest a rather close link between these two polluting substances. In particular, the linear coefficient for direct correlation of the values in question turned out to be 0.78 ± 0.06. Calculating the regression equation indicated that the correlation dependence between PHs (x_1) and total OCPs (y_1) is described by a linear equation of the type:

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[11] The translation of this phrase - 'adekvatnost kontsentratsii DDT' - is uncertain. While 'adekvatnost' in a mathematical context may be translated as 'adequacy', it can also be translated as 'equivalence' or 'goodness of fit'. The precise meaning here is not known. (Tr.)

$$y_1 = 25x_1 - 7,$$

where x_1 represents petroleum hydrocarbons, mg/l;

y_1 represents total OCPs, ng/l.

Concentrations of PCBs in surface waters of the Northwest Atlantic during the period of most intensive use of these substances were very high – from 50 to 4150 ng/l, and in the pelagic division of the North Atlantic from 10 to 200 ng/l (IDOE, 1972). According to data of American scientists (Duce et al., 1973), the maximum concentrations of PCBs in surface waters near the coast of Florida reached 100 μg/l, and significant ΣDDT concentrations were also recorded in this area (Hansen, Wilson, 1970), which was apparently the reason for the destruction of the shrimp population and for the halt in the traditional shellfish hunt in this region, since the maximum allowable concentrations of PCBs in sea water as recommended by experts of NAS-NAE (USA) are 2 ng/l (Karim, 1976). In the early 1970's PCB levels in surface waters of the North Atlantic stood on average at 25-41 ng/l, with maximum concentrations reaching 150 ng/l (Harvey, 1972; Harvey et al., 1973; Osterberg, Keskes, 1977). But by the summer of 1972, after use of these substances had ceased, the average PCB level in surface waters in the open portion of the Atlantic Ocean was 35 ng/l, and 10 ng/l at a depth of 200 m (Harvey et al., 1973). In 1973 PCB concentrations in surface waters of the North Atlantic declined to 1-2 ng/l, and in 1974 reached 0.8 ng/l (Harvey, Steinhauer, 1974a; Osterberg, Keskes, 1977). Results of other research conducted at this same time indicated that average 27/ concentrations of residual amounts of PCBs in surface waters of the North Atlantic, although they had decreased as compared to the late 1960's, nevertheless remained dangerously high for the ecology of this region – 33 ng/l, and 22 ng/l at the 100-200 m depth (Bidleman, Olney, 1974). Soviet scientists have also noted a marked drop in concentrations of chlorinated hydrocarbons in Atlantic Ocean waters in recent years (Kirillova, Orlova, 1979; Orlova, 1982, 1985).

Numerous studies have confirmed that waters of the Baltic Sea and North Sea have high levels of contamination due to chlorinated hydrocarbons (ICES, 1974, 1975; Portmann, 1975; Fischer, 1977).

During the 70's up to 300 t of DDT entered surface waters of the North Sea every year (Vrochinskii, 1976). Rivers of Great Britain carried 20-25 million t of contaminants into coastal waters (Romanova, Martynov, 1976). In the 60's, according to J. Robinson et al. (1967), ΣDDT concentrations in the coastal waters of Great Britain amounted to more than 1 ng/l. By the end of the 70's, thanks to measures undertaken to prevent water pollution (Sauers, 1980), the ΣDDT concentration in coastal surface waters was below the threshold of detectability – less than 0.01 ng/l (maximum concentration 0.03 ng/l), and the PCB concentration fluctuated between 0.2 and 15 ng/l (Dawson, Riley, 1977).

Along the coastal region of Sweden, in the vicinity of industrial and domestic effluent runoff (Stockholm region), the PCB concentration by the late 60's had reached 1350 ng/l, and the ΣDDT 100 ng/l (Ahling, Jensen, 1970). Following the ban on the use of persistent chlorinated hydrocarbons in the majority of West European countries the PCB level in the open part of the North Sea did not exceed 1 ng/l (Portmann, 1975b; Ahnoff, Josefson, 1975; Ahnoff et al., 1979), and the PCB concentration in coastal regions also declined to a level not exceeding 50 ng/l (Rygg, Bokn, 1976). In recent years Sweden has tightened controls over industrial effluents, especially their levels of chlorinated hydrocarbons, with the result that fishing has again been authorized in a number of areas. The Swedish Commission on Water Control and the Swedish Laboratory for the Study of Water and Air Pollution have been set up, and a number of legal measures have been developed to regulate chlorinated 28/ hydrocarbon pollution in the sea (Leif, 1975; Anders, 1975).

During scientific expeditions carried out by us in the fall-winter period of 1979, γ-HCH concentrations in the North Sea did not exceed 10 ng/l, and the ΣDDT level in surface waters varied from 0 to 49 ng/l. In the spring of 1980 lower concentrations of chlorinated hydrocarbons were recorded in this region: the average γ-HCH concentration stood at 7 ng/l, and the ΣDDT at 26 ng/l. In the Kattegat and Skagerrak in 1979 DDT was discovered in trace concentrations. Concentrations of α-HCH and lindane averaged 13.3 and 7 ng/l, respectively. In the spring of 1980 higher average concentrations of OCPs were recorded in these straits: ΣDDT – 17.7, α-HCH – 14.7, lindane – 7 ng/l.

For the open portion of the Baltic Sea a tendency was noted for a decrease in concentrations of residual OCPs in surface waters during the period 1976 - 1980 (1976: 0.8 - 10.5; 1977: 0.9 - 4.2; 1978: 0.05 - 2.9; 1979: 0.05 - 0.0; 1980: 0 - 0.5 ng/l) and a stabilization in average PCB concentrations in sea water (Stadler, Ziebarth, 1976; Stadler, 1977; Roots et al., 1977; Itra, 1978; Brugmann et al., 1980; Gaul, Ziebarth, 1980; Roots, 1981, 1982; Roots, Peikre, 1981; Roots, 1983). In 1981, in the outlet from the Kurskii zaliv [Kurskii Bay] and in the vicinity of Arkonskaya vpadina [Arkonskaya Depression] ΣDDT of 10 - 11 ng/l was discovered, and DDE was found in the outlet from the Gulf of Finland in a concentration of 4.3 ng/l; lindane was discovered everywhere in background concentrations. The highest concentrations of lindane were measured in the area of Gotland Island – up to 46 ng/l (Shukite, 1985). As compared with 1978

(Chernyak et al., 1979), DDT concentrations in bottom deposits had decreased from 8 ng/g of dry matter to total absence, while γ-HCH concentrations had risen from 9 to 41 ng/g dry matter, indicating a continuous input of this type of chlorinated hydrocarbons (Shukite, 1985). In the western part of the Baltic Sea and in the Sund Strait[12], in the spring of 1980, we recorded higher OCP concentrations than in the open parts of the sea: the average DDT concentration in surface waters equalled 26 ng/l and varied between 15 and 48 ng/l. Average concentrations of α- and γ-HCH in surface waters in this region amounted to 9.5 and 9.2 ng/l, respectively (Savinova, Ugryumova, 1981).

The dynamics in levels of residual chlorinated hydrocarbons in surface 29/ waters of the Norwegian and Barents seas are closely linked to peculiarities in their hydrologic regime and to the possible transport of contaminants by Atlantic currents and their continuations (Simonov et al., 1974; Savinova et al., 1982), and also to the pathways of bioaccumulation and transport through trophic chains (Savinova, 1986). In the waters of the North Cape Current, at the entrance to the Barents Sea, average concentrations of residual organochlorine compounds, in ng/l, were: p,p'-DDT up to 80, o,p'-DDT up to 20, lindane up to 12.5 (Savinova et al., 1982). In the waters of the Barents Sea the largest concentration of total DDT was detected in Atlantic waters of the Murmansk coastal current and amounted to 50 ng/l; in the open part of the sea and eastern coastal region, as the influence of Atlantic waters receded, the OCP concentrations were significantly lower – at background levels (Savinova et al., 1981).

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[12] This may mean the 'Øre Sund' (Tr.).

In the Norwegian Sea, according to Yu.A. Izrael et al. (1980), OCPs have been found both in the surface layer (0 - 100 m) as well as at a depth of 800 m. In studies carried out by us on the seasonal dynamics of OCP contamination in waters of the Norwegian Sea (1979 - 1980) higher concentrations of toxicants were recorded in the fall-winter period. Maximum ΣDDT concentrations in surface water samples from the Norwegian Sea reached 53 ng/l in the spring versus 109 ng/l in the winter. The higher local concentrations of OCPs in the fall-winter period can apparently be explained by the large influx of warm Atlantic waters, which bring the contamination with them. The concentrations of α-HCH and lindane in surface water samples from the Norwegian Sea in the winter and spring time frame are characterized by approximately equal values, varying in winter samples from 3 to 26 and from 4 to 11 ng/l, and from 0 to 25 and 0 to 16 ng/l in spring samples. Maximum concentrations of α-HCH and lindane were determined at the mouth of the Geta-Elv River, apparently related to shore runoff, and reached 30 ng/l for α-HCH and 15 ng/l for lindane. Elevated concentrations of all organochlorine compounds were discovered in a zone of converging waters of varying provenances (Medvezhii Island region). Figure 4 offers a sketch map showing pollution of surface waters in the 30/ Norwegian Sea in the fall-winter period.

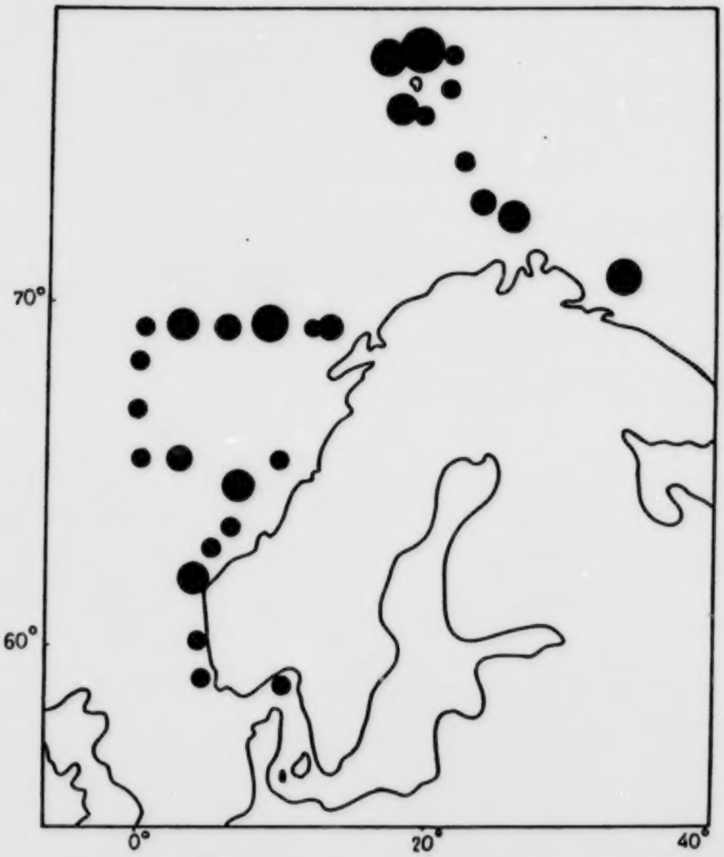

Рис.4. Карта-схема степени загрязненности поверхностных вод Норвежского моря п,п -ДДТ, ноябрь-декабрь I979 г. (нг/л).

● - 30, ● - 30-40, ● - 40-50, ● - 50-60, ● - 60-70, ● - 70-80, ● - 80-I00.

Figure 4. Sketch map showing degree of p,p'-DDT pollution of surface waters in the Norwegian Sea, November-December 1979 (ng/l).

Studies conducted in April-June 1981 have shown the presence of OCPs at all water levels in the Norwegian Sea and have confirmed that maximum 31/ concentrations are confined to flows of the principal Atlantic currents. At a depth of 1000 m the concentration of lindane reached 8 - 10 ng/l, and ΣDDT 0 - 16 ng/l. The fact, noted by us during our research, that the areas of greatest chlorinated hydrocarbon pollution coincided with high productivity zones in the

Norwegian Sea and western part of the Barents Sea, along with slower processes of natural biodegradation due to low average annual water temperatures (Jensen, Olsson, 1973), are creating a threat to the viability of hydrobionts in this region.

3.2 Petroleum Hydrocarbons

In the Barents Sea elevated PH concentrations have been observed in frontal zones and areas where fishing fleets operate. Thus, for example, the highest concentrations (up to 6.2 mg/l) were detected in a convergence zone (Norina, 1975). Oil and petroleum products are transported into the Barents Sea by the Atlantic current system. It is estimated that the North Atlantic current system transports approximately one million t of petroleum hydrocarbons annually (Nesterova, Simonov, 1979). In the open part of the Barents Sea maximum concentrations of PHs are observed in Atlantic waters, especially in the North Cape Current, and decrease in an easterly direction. In the western part of the sea the PH level in the 0 - 50 m layer averages 1.6, versus 0.7 mg/l in the eastern part (Norina, 1975). In coastal regions, according to K.N. Bogdanova et al. (1977), the PH level averages 2.27 mg/l, with concentrations fluctuating between 0.01 and 30.8 mg/l, while in the open part of the Barents Sea the PH concentration during the period of research equalled 0.04 mg/l (0.01 - 0.48 mg/l).

In the years 1975 - 1978 Norwegian scientists conducted regular observations of oil pollution in the North Sea, in a region of intensive oil production on the shelf, and in the Barents Sea. The average concentration of oil aggregates in surface waters of the North Sea amounted to 0.2 mg/m^2, and $^2/_3$ of the samples contained tar balls. In the Barents Sea the concentration of tar balls was significantly lower and scarcely attained 0.1 mg/m^2, while they were found in $^1/_3$ of the samples (Heyerdahl, 1978). According to Butler (1975) 32/ the concentration of oil aggregates in surface waters of the Northeast Atlantic reached 2.4 mg/m^2. From all of the samples collected in the years 1977 - 1979 from surface waters of the North Atlantic (Mikhailov, 1985), oil aggregates were noted in 65%. Their levels varied widely from 0.1 to 218.5 mg/m^2. In 9% of the samples their level stood at less than 0.1 mg/m^2, in a range of 0.1 to 1.0 mg/m^2 in 52.5%, between 1.0 and 5.0 mg/m^2 in 31.4%, and at more than 5.0 mg/m^2 in 7.1%. The average concentration of oil aggregates in this region (based on three years of monitoring data) was 0.4 mg/m^2 and agrees with the concentration previously cited from Kohnke (1978). Also of interest is a determination of the general mass of oil aggregates found in the North Atlantic surface layer. Data on their levels is extremely sparse, although the data was used by English scientists to calculate the general mass of oil aggregates – $2.7 \cdot 10^4$ t in 1971 (Morris, 1971), and $7.0 \cdot 10^4$ t in 1973 (Morris, Butler, 1973). Other calculations place the overall level of oil aggregates in the North Atlantic at $2.4 \cdot 10^4$ t (McAnliff, 1976). According to Soviet researchers (Mikhailov, 1985) the surface waters of the North Atlantic as a whole contain approximately $20 \cdot 10^3$ t of oil aggregates, which approximates the figures cited above from Kohnke (1979). Observations made in the North Atlantic have shown that the highest PH pollution is in shelf waters of continental and insular regions, where the level of these contaminants varies between 0.05 and 0.70 mg/l (Butler, 1973). Canadian researchers measured oil levels in the water column in the Gulf of Saint Lawrence over a period of nine years. Despite significant fluctuations in concentrations from year to year, an overall decrease in background levels was traced over the decade. In 1979 the average level stood at 0.4 μg/l, corresponding to the unpolluted waters of eastern and Arctic Canada where precipitation of oil products from the atmosphere is the principal source of entry into the sea (Levy, 1985).

Research on distribution of PHs in waters of the North Atlantic carried out in the 70's by scientists of GOIN (Simonov et al., 1974; Simonov, 1978) clarified the role of the basic circulatory systems and of relatively stagnant zones of oceans and seas in the transport and accumulation of pollutants. The greatest 33/ concentrations were recorded in coastal zones and in extensive, relatively immobile ocean regions into which they are carried by the main current systems. Thus the Gulfstream of the North Atlantic Current, saturated with contaminants near the coasts of Europe and North America, has several discharge zones, including the Sargasso, Norwegian and Barents seas. These discharge zones, including the Arctic region, then become accumulators of harmful substances.The transport of pollutants takes place primarily in peripheral zones of circulatory systems where they are concentrated by the transverse component of the velocity of the current. Significant concentration of PHs has also been discovered in the SML, which is attributed to their physical and chemical properties, especially to a somewhat lower density as compared to the density of sea water and to their low degree of solubility (Simonov, Mikhailov, 1979)

In the North Atlantic, PH concentrations in the SML vary within a wide range (from 0.2 to 15 mg/l) and are characterized by substantial spatial disuniformity caused by various circulatory systems, frontal zones, and by the uneven input of PHs (Mikhailov, 1985). The North Atlantic Current system has a level of PH pollution in the SML from 0.1 to 1.0 mg/l. The average concentration of PHs throughout the entire Northeast Atlantic in 1978 stood at 0.58 mg/l. In this region a decrease is noted in PH concentrations in the SML at greater distance from the shelf zone of Ireland, which can be most clearly observed in the northern part of the region. This is linked to the dispersal in the ocean of the more polluted surface waters issuing from the Irish Sea (more than 2.0 mg/l). Waters washing the shelf of southeastern Iceland, which are by provenance transformed waters of the North Atlantic Current, characteristically possess significant PH concentrations in the SML (from 1.4 to 2.37 mg/l). A characteristic trait of PH spatial distribution in the SML in regions between Iceland, the Faeroe Islands and Norway is an increase in PH concentrations closer to the shelf of Norway and the North Sea. Soviet scientists estimate that, at the present time, approximately $1.2 \cdot 10^2$ t of oil and petroleum products are entering the North Atlantic as a result of anthropogenic activity (Orlova, 1977; Smagin, Rachkov, 1977).

The level of dissolved/emulsified PH fractions in surface waters in the sub- 34/ Arctic region of the North Atlantic in 1977 varied between 0 and 0.06 mg/l (Mikhailov, 1985), which agrees with the data from scientists of AANII [Arctic and Antarctic Scientific Research Institute] (Smagin, Rachkov, 1977) – from 0 to 0.04 mg/l. The least polluted were sub-Arctic waters north of 50° N.L., where no PHs were detected in over 70% of samples.

The vertical distribution of PHs in waters of the North Atlantic shows a tendency toward lower concentrations with depth. Primary pollution is concentrated in the upper 10-metre layer of water, with PHs virtually absent at depths of 100 m or more. Another distinctive feature of the vertical distribution of PHs is their tendency to accumulate in a layer of discontinuity in density. In all probability, the density discontinuity layer in a number of areas of the North Atlantic is the lower boundary of PH distribution and serves as a kind of PH "filter" (Mikhailov, 1985).

Approximately 50% of the oil and petroleum products entering the Baltic Sea comes from constant sources (river, domestic and industrial runoff), 20% from the loading and unloading of tankers, and 10% from accidental spills and other sources. The total input of oil is estimated to be $50 \cdot 10^3$ t/yr (Jensen, Hansen, 1982). Melvasalo (1980) made a determination of the overall influx of petroleum hydrocarbons into the Baltic Sea in 1980: $(50\text{-}100) \cdot 10^3$ t/yr, which accounts for 1.4 - 2.8 % of all the oil and petroleum products entering the marine environment. According to Shukite (1985) PH pollution of waters of the Baltic Sea is not significant and has now declined 8-fold as compared with 1978. Characteristically, the distribution of PHs both in the surface layer and in the total water mass in the course of a year is highly uniform.

In 1981, for example, PH concentrations in the surface layer over the entire sea fluctuated between 0 and 0.02 mg/l, and concentrations up to 0.04 mg/l were observed only in the southeastern part of the sea. Concentrations of PHs somewhat above background levels (up to 0.02 - 0.03 mg/l) were observed in the fall, especially, in the surface horizon on the approach to the Denmark Strait and the the Gulf of Finland, and in the seafloor horizon in regions of the Gotland and Bornholm depressions. Further evidence for accumulation of PHs in the vicinity of deep-water depressions is provided by the higher PH levels in bottom deposits, especially in the Gotland Depression region: up to 0.15 35/
mg/g of dry matter (compared with an average of 0.04 mg/g of dry matter for the [entire] sea).

Chapter 4. POLLUTANT LEVELS IN NORTH SEA HYDROBIONTS

4.1 Plankton

One of the main accumulators of contaminating substances is phytoplankton, and it plays the central role in the distribution of toxicants in aquatic ecosystems (Fig. 2).

In the opinion of S.A. Patin (1977) concentrations of contaminants of anthropogenic origin recorded in sea water and plankton are capable of significantly retarding the rates of formation of organic substances in the world's oceans: a global reduction of 10% in primary production should bring with it a corresponding reduction in the rate of production at other trophic levels as well, all the way to the necton, where these losses already total tens of millions of tonnes, including millions of tonnes of commercial fish annually.

The chain "diatom – zooplankton" may be one of the most extensive chains in the transport of chlorinated hydrocarbons since the presence of fatty inclusions in diatoms results in a high level of accumulation of these contaminants. It has been shown experimentally that the coefficient of accumulation of DDT in the diatom *Cyclotella* sp. may reach 37 000 units, and 32 000 units in *Skeletonema* sp. (Ernst, 1975). There has been little study of the consequences of this process, which appear to manifest themselves in disruptions of interrelationships in marine ecosystems, the death of commercial organisms, the replacement of some species with others, etc. Often, these biological consequences are not obvious.

The mechanism by which chlorinated hydrocarbons accumulate in the algae is not as yet altogether clear. It seems that some sort of equilibrium is 36/

established between concentrations of chlorinated hydrocarbons in water and in phytoplankton. Concentrations of DDT and PCBs in the plankton attain equilibrium with a concentration in the environment in the course of several seconds or hours, depending on the species of alga and on the physico-chemical properties of the toxicants (Sodergren, 1968; Biggs et al., 1980).

Measurements of the fluorescence of chlorophyll "a" have shown that PCBs accumulate in photosynthetically active centres sensitive to the effect of these compounds (Harding, Phillips, 1978b).

Table 4 presents data, obtained from a small number of analyses, characterizing the accumulation of chlorinated hydrocarbons in plankton.

As can be seen from Table 4, the level of residual chlorinated hydrocarbons in plankton of northern seas is approximately uniform.

The coefficients computed for accumulation of DDT type compounds for mixed plankton samples from various regions of the North Sea have shown that the highest coefficients of accumulation are confined to regions of intense oil and gas production on the shelf (Fig. 1). According to Beyer and Rainter (1977), oil losses from drilling platforms during production on the shelf amount to 72 g per tonne of oil produced. DDT has a higher degree of solubility in oil (10^6 times greater than in water), and being dissolved in oil can penetrate the cell membrane more actively than if suspended or dissolved in water. It cannot be excluded that it is precisely the higher rate of penetration of cell membranes by DDT dissolved in oil that explains the higher coefficients of accumulation of DDT by plankton in this region.

Data on the level of petroleum hydrocarbons in the plankton of northern seas is virtually nonexistent, due to the analytical difficulties of determination and identification. According to Bogdanova et al. (1987) zooplankton in coastal areas of the Barents Sea accumulate up to 3.5% of the petroleum hydrocarbons in sea water, and for this reason it has been suggested that zooplankton be viewed as an indicator for petroleum hydrocarbon pollution of water in the Barents Sea.

Table 4

Residual levels of chlorinated hydrocarbons in phytoplankton, zooplankton and mixed plankton (mg/kg)

	ΣDDT		PCBs		
Locality, type of sample	Wet weight	Lipid weight	Weight weight	Lipid weight	Source
1	2	3	4	5	6
North Atlantic, phytoplankton	0.004	–	–	–	Williams, Holden, 1973
Open part of Atlantic, phytoplankton	(0.0002 - 0.0005)	–	–	–	Harvey et al., 1973
Baltic Sea, Gulf of Bothnia, phytoplankton	–	–	(0.012 - 0.17)	–	Kihlström, Berglund, 1978
Baltic Sea, mixed plankton	–	–	–	18.0 (3.0 - 35.0)	Jensen et al., 1972b
Baltic Sea, mixed plankton, 1973	–	–	0.18 (0.004 - 0.75)	25.0 (4.0 - 77.0)	Linko et al., 1974
Baltic Sea, mixed plankton, 1976	–	–	0.23 (0.04 - 0.72)	28.0 (4.0 - 66.0)	Linko et al., 1979

1	2	3	4	5	6
Baltic Sea, mixed plankton, 1974	–	–	0.37	0.38	Linko et al., 1979
Baltic Sea, zooplankton, 1972	(0.002 - 1.5)	–	–	(0.009 - 6.5)	Miettinen, Hattula, 1979
Baltic Sea, zooplankton, 1978	(0.012 - 0.19)	–	–	–	Roots, Peikre, 1981
North Sea, coast of England, microzooplankton	0.03	–	–	–	Robinson et al., 1967
Kattegat, mixed plankton, 1980	0.043	–	–	–	Savinova, 1982
Skagerrak, mixed plankton, 1980	0.047 (0.01 - 0.09)	–	–	–	
North Sea, mixed plankton, 1980	0.025 (0.002 - 0.09)	–	–	–	

(Table 4 continued)

1	2	3	4	5	6
Norwegian Sea, mixed plankton, 1980	0.013 (0.001 - 0.09)	–	–	–	Savinova, 1982
Atlantic, mixed plankton	(0.003 - 0.1)	–	0.2	2.3	Harvey et al., 1974; Williams, Holden, 1973
North Atlantic, zooplankton, 1973 - 1974	–	(0.1 - 2.7)	–	–	Sameoto et al., 1975
North Atlantic, mixed plankton	–	–	31.0	–	Ware, Addison, 1973
Northeast Atlantic, mixed plankton	–	–	(0.01 - 0.12)	(0.1 - 5.5)	Holden, 1973
Northeast Atlantic, mixed plankton, 1971	–	–	0.6 (0.01 - 0.11	–	Holden, 1972
Atlantic zooplankton	–	0.095	–	–	Harvey et al., 1974

(Table 4 concluded) 40/

1	2	3	4	5	6
Northwest Atlantic, mixed plankton	–	–	0.15 (0.01 - 0.3)	40.0 (2.4 - 260.0)	Risebrough et al., 1972
South Atlantic, mixed plankton	–	–	0.20 (0.02 - 0.64)	40.0 (7.0 - 120.0)	
South Atlantic, zooplankton	(0.004 - 0.064	–	–	–	Lukowski, 1978
Gulf of Mexico, mixed plankton	–	–	0.095 (0.003 - 1.06)	–	Giam et al., 1973
Gulf of Mexico, mixed plankton	–	–	0.084 (0.04 - 0.157)	–	Baird et al., 1975

NOTE: a dash indicates no determination made.

4.2 Benthos

Because of the known capacity of chlorinated hydrocarbons to concentrate in suspended (sedimentary) material and to be removed from water by sedimentation, significant concentrations of these substances accumulate in bottom sediments and, consequently, elevated levels of toxins are to be expected in benthic organisms. This has been confirmed by studies carried out on the degree of contamination of benthic organisms in the North and Northeast Atlantic (ICES, 1974, 1977; Portmann, 1979; Murray, 1979, 1982). Experimental studies conducted by Nimmo et al. (1971a) have shown that PCBs (Aroclor 1254) enters the estuarine food chains from bottom sediments. Many invertebrates are capable of effectively accumulating chlorinated hydrocarbons from sediments with a large quantity of organic matter (Odum et al., 1969).

Of great significance in the accumulation of chlorinated hydrocarbons by marine invertebrates are the processes of feeding and respiration (Butler, 1966, 1969; Goldberg, 1972). Table 5 presents results of experiments to study the ability of certain invertebrate species to accumulate chlorinated hydrocarbons. As can be seen from Table 5, aquatic invertebrates accumulate chlorinated hydrocarbons in significant quantities and in a comparatively short period of time.

Professor E. Goldberg (1972), a noted specialist in the study of marine pollution, has expressed the opinion that mollusk-filtrators, which can rapidly alter concentrations of contaminants in their body in relation to changes in [contaminant] levels in the environment, should be regarded as true indicators of pollution.

The general level of chlorinated hydrocarbons in bivalve mollusks from various regions of the world's oceans is presented in tables 6 and 7.

Table 5   42/

Experimental accumulation of chlorinated hydrocarbons by various species of benthic organisms

Species	Concentration (μg/l)	Coefficient of accumulation (thousands)	Exposure	Source
		DDT		
Mercenaria mercenaria	1.0	6	1 wk	Butler, 1966
Mya arenaria	0.1	8.8	5 days	Butler, 1971
Crassostrea gigas	1.0	20	7 days	Butler, 1966
Penaeus duorarum	0.14	1.5	3 wks	Nimmo et al., 1970
P. duorarum	0.1	10.6	2 wks	
P. duorarum	1.0	38	2 wks	Hansen, Wilson, 1970
		Aroclor		
Palaemonetes pugio	0.62	2.07	1 wk	Duke, Dumas, 1974
P. pugio	0.62	25.6	5 wks	
Penaeus duorarum	2.5	1.8	2 days	Nimmo et al., 1971b

Table 6 43/

Level of residual chlorinated hydrocarbons in bivalve mollusks
(µg/kg wet weight)

Locality	PCBs	ΣDDT	Source
1	2	3	4
		Mytilus edulis	
Coast of California, 1965 - 1966	–	19.0 - 34.0	Risebrough et al., 1967
Baltic Sea, 1966 - 1968	30.0 - 84.0	20.0 - 40.0	Jensen et al., 1969
Baltic Sea, 1966 - 1967	97.0	40.0	
Coast of the Netherlands	–	21.0	Holden, 1970
Coast of Southern Europe	–	183.0	
Coast of Great Britain	–	43.0	
Coast of Northern Europe	–	20.0	
Coast of Canada	–	69.0	
Coast of Canada		73.0 (10.0 - 170.0)	Sprague, Duffy, 1971
Coast of Canada, 1970	140.0	20.0	Zitko, 1971
North Sea, 1965 - 1968	600.0 - 1100.0	100.0 - 250.0	Koeman, Van Genderen, 1972
Coast of Canada, 1971 - 1972	0.023 (0.09 - 0.05)	0.015 (0.01 - 0.02)	Sims et al., 1977
Coast of Sweden, 1972	95.0	13.0	ICES, 1974
Coast of FRG, 1972	90.0	25.0	
North Sea, coast of Norway, 1972	30.0	25.0	

(Table 6 concluded)

1	2	3	4
North Sea, coast of the Netherlands, 1973	273.0	9.0	Ten Berge, Hillebrand, 1974
North Sea	100.0 - 650.0	3.0 - 5.0	Qurijns et al., 1979
North Sea, coast of Great Britain, 1970 - 1973	65.0	13.0	Portmann, 1979
North Sea, coast of Great Britain, 1974	160.0	20.0	Murray, 1979
", 1975	80.0	12.0	
", 1975	–	4.0 - 237.0	Butler, 1973
Astarte borealis *Macoma baltica* *Modiolus modiolus*			
Barents Sea, coast of Eastern Murman	Not detected	Not detected	Savinova et al., 1981

NOTE: dash indicates no determination made.

Among the basic criteria imposed on indicator organisms are, first of all, massiveness, cosmopolitanism and sedentariness, and, secondly, the capacity to accumulate and concentrate pollutants while maintaining the principal signs of vitality and genetic stability at relatively high environmental concentrations. It 44/ is also essential that indicator organisms be easily accessible for collection and that they have a sufficiently long life span so that observations can encompass a number of years.

Mollusks of the class Bivalvae generally satisfy the requirements listed. Among hydrobionts they possess one of the highest coefficients of accumulation thanks to their ability to filter huge quantities of water and to concentrate substances dissolved in it.

Many researchers feel that species of the genus *Mytilus* are especially promising as indicator organisms (Butler, 1969b; Goldberg, 1975; Risebrough et al., 1976a; Burdin et al., 1979).

A program has been underway in the USA since 1976 to monitor the level of pollution in sea water by analyzing tissues of oysters and mussels from 107 45/ standard stations along the eastern and western shores of the Gulf of Mexico (Bayne, 1978). In Great Britain a program of pollution monitoring since 1970 has used both mussels and other benthic organisms: *Modiolus modiolus, Chlamys opercularis, Pecten maximus, Buccinum undatum, Cancer pagurus, Crangon crangon, Pandalus montague, P. Borealis, Eupagurus bernhardus* (Portmann, 1979; Murray, 1979, 1981, 1982).

Results of the research have shown it is possible to use gastropod mollusks, especially the genus *Buccinum*, as indicator organisms (Cole, 1979). As our studies indicated, no residual chlorinated hydrocarbons were detected in *B. undatum* in coastal waters of the Eastern Murman region while the muscles of mollusks of this species from the western part of the Barents Sea coast – a more polluted region – had average levels of 0.047 mg/kg of wet weight. In this way it was shown that using mollusks of this species to monitor pollution is also promising for the Barents Sea.

In experimental research on accumulation of chlorinated hydrocarbons in organs and tissues of crabs it was established that the maximum quantities of toxicants accumulate in the hepatopancreas, followed by the brain, thoracic ganglia, hemolymph, gills, and muscles. An almost complete absence of accumulation was noted in the heart and blood. It was concluded that chlorinated hydrocarbons are absorbed by the gills and pass into the hepatopancreas along with the hemolymph (toxicants accumulate in the hemolymph 5 minutes, and in the hepatopancreas 15 min after exposure) (Sheridan, 1975).

Tables 8 and 9 show general levels of chlorinated hydrocarbons in various species of shellfish.

In *Pandalus borealis* shrimp from the western part of the Barents Sea the following concentrations of residual chlorinated hydrocarbons have been measured: ΣDDT – 6.9 (0.3 - 14.7), PCBs – 7.0 (1.2 - 19.8) and lindane – 4.8 (0.5 - 20.6) μg/kg. The level of chlorinated hydrocarbon contamination in shrimp reflects the general tendency in accumulation dynamics for these toxicants, a characteristic of which is a decrease in recent years in DDT accumulation and a predominant accumulation of PCBs.

Table 7 46/

Level of residual chlorinated hydrocarbons in marine scallops
(mg/kg of wet wt)

Scallop species	Report area	Type of sample	PCBs	ΣDDT	Source
Placopecten	Coastal waters of Canada	Muscles	–	0.03 (0.01 - 0.09)	Sprague, Duffy, 1971
P. magellanus	"	Muscles	0.018 (0.005 - 0.051)	0.03 (0 - 0.01)	Sims et al., 1977
Chlamys opercularis	Coastal waters of Great Britain, 1974	Muscles	(0.02 - 0.05)	(0.003 - 0.007)	Ernst et al., 1976
	"	Muscles	0.05	0.004	
		Gonads	0.01	0.003	
	" 1975	Muscles	0.01	0.003	
Pecten maximus	" 1975	Muscles	0.01	0.003	
		Gonads	0.02	0.003	
Chlamys islandica	Barents Sea, 1979	Muscles	–	Not detected	Savinova et al., 1981
		Gonads	–	0.012	

NOTE: the dash indicates no determination made.

Table 8 47/

Level of residual chlorinated hydrocarbons in various species of crab
(mg/kg of wet wt)

Species of crab	Locality	PCBs	ΣDDT	Source
Una minax	Atlantic ocean	0.45 - 1.5	–	Nimmo et al., 1971b
Minula tenuimana	Skagerrak	0.01 - 0.03	0.004 - 0.008	Eder et al., 1976
Cancer pagurus	North Sea, 1970 - 1973	2.3	0.137	Portmann, 1979
C. pagurus	Same, 1974	0.44	0.576	Murray, 1979
Eupagurus bernhardus	Same, 1974	0.16	0.095	
C. pagurus	Same, 1975	0.92	0.100	
E. bernhardus	Same, 1975	0.75	0.065	
Cancer irroratus	Coast of Canada, 1971 - 1972	0.024	0.024	Murray, 1979
Gergon quinguedens	Coast of Canada, 1971 - 1972	0.036	0.061	Sims et al., 1977
	Same, 1971 - 1972	0.027	0.018	
Hyas araneus	Barents Sea, Eastern Murman	Not detected	Not detected	Savinova et al., 1981
H. araneus	Western part of Barents Sea	–	0.01 - 0.09	

NOTE: the dash indicates no determination made.

Table 9 48/

Level of PCBs and DDT in shrimp
(mg/kg of wet wt)

Species of shrimp	Locality	PCBs	ΣDDT	Source
Penaeus sp.	Coastal waters of Florida	–	0.160	Woodwell et al., 1967
P. duorarum	Same	14.0	43.0 (10.0 - 140.0)	Nimmo et al., 1971b
P. setiferus	Pacific Ocean	–	0.0 - 0.2	Keisler et al., 1973
P. aztecus	Same	–	0.006 - 2.5	Butler, 1973
Pandalus borealis	Atlantic Ocean, Georges Bank	0.36	0.007	Harvey et al., 1974
P. borealis	Denmark Strait	0.018	0.001	
Crangon crangon	Southwestern North Sea	0.083	0.003	Ten Berge, Hillebrand, 1974
P. borealis	Skagerrak	0.01 - 0.02	0.001 - 0.003	Eder et al., 1976
P. borealis	Barents Sea, Eastern Murman	–	0.001	Savinova et al., 1981
P. borealis	Barents Sea, shore of Medvezhii Island	0.007	0.006	Author's data
P. borealis	Coast of Canada	0.045	0.003	Sims et al., 1977
C. crangon	North Sea, coast of FRG	–	0.002	Goerke et al., 1979
C. crangon	North Sea, coast of Great Britain	0.093	0.001	Portmann, 1979
P. montague	Same, 1970 - 1973	0.11 0.04	0.009 0.005	
P. montague	Same, 1974			Murray, 1979
P. montague	Same, 1975	0.10	0.013	
P. montague	Barents Sea, Eastern Murman	–	0.005	Savinova et al., 1981

NOTE: the dash indicates no determination made.

The available data indicates a negative correlation between shrimp weight and level of lindane in the shrimp, and a ΣDDT with coefficients of linear correlation of –0.59 and –0.78, respectively. No correlation with PCB level was detected. The relationship between the ΣDDT accumulation level and the weight of shrimp is shown in Figure 5. This relationship is approximated by the 49/ function $y = \frac{20.118}{W} + 0.612$, with correlation coefficient $r = -0.90$.

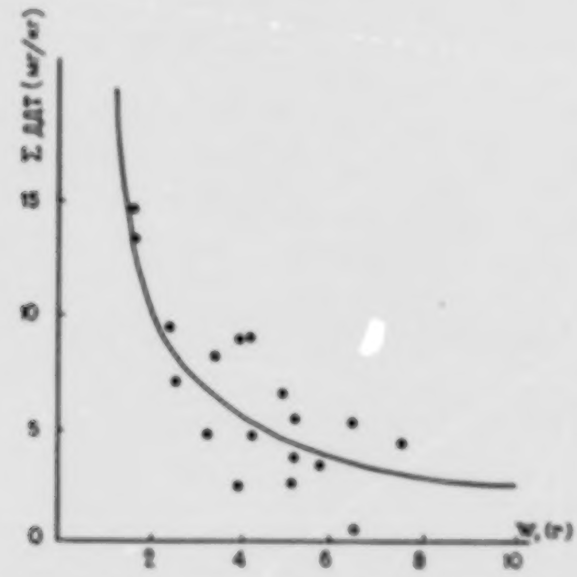

Рис.5. График зависимости между весом креветок Pandalus borealis и содержанием в них ΣДДТ.

$y = \frac{20.118}{W} + 0.612, \; r = -0.90.$

Figure 5. Relationship between weight and ΣDDT content in the shrimp *Pandalus borealis*. [Horizontal axis: weight (g); vertical axis: ΣDDT (mg/kg).]

These results show it is essential to take into consideration the size of the shrimp when using them as indicator organisms, when conducting research on xenobiotic detoxification processes, and also during commercial fishing. The latter is particularly important since 1 kg of small shrimp caught contains a significantly larger quantity of contaminants as compared with large shrimp (Savinov, Savinova, 1986).

Table 9 shows data on residual levels of chlorinated hydrocarbons in shrimp from various regions of the world's oceans.

There has been notably less research on accumulation levels of chlorinated hydrocarbons in echinoderms. There has been study only on the ΣDDT level in the gonads of sea urchins of the genera *Eshinus* and *Strongylocentrotus* – 0.05 and 0.005 mg/kg, respectively (Robinson et al., 1967; Risebrough et al., 1967) and 10 to 78 mg/kg in starfish of the genera *Pisaster, Patria* and *Acanthaster* (Risebrough et al., 1967; McClosky, Deubert, 1973). In our research no residual chlorinated hydrocarbons were detected in the gonads of the sea urchin *Strongylocentrotus droebachiensis* from coastal areas of the Eastern Murman.

Due to analytical difficulties in the determination and identification of petroleum hydrocarbons in the organs and tissues of hydrobionts there is 50/ extremely limited data on the level of bioaccumulation of these contaminants by benthic animals in northern seas.

It has been established experimentally that dissolved hydrocarbons are taken up by the gills of the mollusk *Mytilus edulis* and transported to other tissues (Lee et al., 1972). It has not been established which of the means of petroleum hydrocarbon uptake in bivalve mollusks is the primary one – ingestion with food or absorption from water – but it can be supposed that the quantity of hydrocarbons removed by the animals from the water they filter to obtain oxygen is an order higher than the quantity obtained from food, and that respiration is the principal route by which the pollutants enter. In some cases a significant quantity of hydrocarbons may enter the body of benthic animals from 51/ polluted bottom sediments (Stegeman, Teal, 1973).

Based on two populations of *Crassostrea virginea* with different lipid levels, it was found that the rate of absorption and degree of hydrocarbon accumulation are related to the quantity of lipids. But this begins to manifest itself following a certain general accumulation of hydrocarbons in the tissues of the mollusks, as has also been observed in the absorption of chlorinated hydrocarbons. So long as the concentration of hydrocarbons in the organism is still rather small, the rate of their accumulation is dependent on the overall mass of the hydrobiont. When the concentration of hydrocarbons in the organism attains a high level, a relationship between the rate of their absorption and lipid level can be determined (Stegeman, Teal, 1973). It is natural that the initial absorption rate should depend on the concentration of petroleum hydrocarbons in the water. At high concentrations of petroleum hydrocarbons the mollusks may remain closed and the rate of absorption of contaminants will be minimal.

Mussel monitoring for chlorinated and petroleum hydrocarbon pollution levels carried out along the coast of Canada (from June, 1983, to October, 1984) determined the presence of contaminants in mussels on the western coast of Canada, on the eastern coast at the entrance to Halifax harbor, and in an unpolluted area in the estuary of the St. Lawrence River far from urbanized zones. On the western coast the highest concentration of alkanes (6 - 32 mg/g of dry wt) and aromatic substances (11 different compounds, 270 - 4600 μg/g of dry wt) were discovered in the tissues of mollusks. On the eastern coast and in the river estuary the petroleum hydrocarbon level in mussels was significantly lower (Hamilton et al., 1987).

In the vicinity of a marine drilling platform in the Norwegian Sea the level of naphthalene, phenanthrene and benzothiophene in the tissues of *Mytilus edulis* attained the highest point – 8 μg/g of wet wt (Grahl-Nielsen, 1987).

Scientists of the University of Alaska studied the level of petroleum hydrocarbons in the mussels *M. edulis* and *M. baltica* collected in a sub-Arctic fjord near the terminal of the Transalaska Pipeline, where discharges of 170 kg of oil per day are permitted. [Levels of] 936 μg/g of dry wt were recorded in 52/
the tissues of *M. edulis*, and 449 μg/g in *M. baltica* (Shaw et al., 1986).

The method of spectral luminosity has made it possible to study the dynamics and distribution in various species of North Atlantic hydrobionts of the group of carcinogenic polycyclic aromatic hydrocarbons. High levels of benz[a]pyrene have been measured in neustonic (Sargasso crab – 4.06 μg/kg of dry wt), pleustonic (Portuguese man-of-war – 27.60 μg/kg) and nectonic organisms (squid – 3.09 μg/kg). Polar and low-polar compounds in integral samples of plankton amounted to 10.0 - 28.0 mg/g of dry plankton. Analysis has indicated a tendency in recent years toward a decrease in benz[a]pyrene concentrations in various organisms in northern seas (Zubakina et al., 1986).

4.3 Fish

A large number of studies have been devoted to levels of chlorinated hydrocarbon concentration in marine fish. Experiments have shown that chlorinated hydrocarbons may enter the fish organism both directly from water – by adsorption on their outer surface and from water through the gills (Holden, 1962; Murphy, 1971; Hamelink et al., 1971) – as well as (mainly) via food (Macek et al., 1970; Macek, Korn, 1970; Grzenda et al., 1970).

Kenaga (1972) feels the data on bioaccumulation of pesticides permits the conclusion that, initially, the accumulation of residual pesticide concentrations is due to adsorption, being frequently related to a high ratio of surface area to mass of the adsorbent, and can lead to a high level of hydrobiont contamination. Subsequently, bioaccumulation takes place as a result of absorption and ingestion with food. Persistent chlorinated hydrocarbons, which possess a high degree of solubility in fats and low solubility in water, can continuously increase the level of accumulation in hydrobionts until such time as an equilibrium is attained by means of toxicant distribution between sea 53/ water and lipids of the organism (Koeman, 1972a; Schneider, 1978, 1982).

The accumulation of chlorinated hydrocarbons by fish depends on the sex, season, and feeding conditions (Perttilä et al., 1982; Addison, 1976, and others). A relationship has been determined between the level of accumulation of DDT and PCBs and the age of marine fish (Cox, 1970; Hansen, Wilson, 1970; Murphy, 1971; Stenersen, Kvalvag, 1972; Jensen et al., 1972a; Bjerk, 1973; Schaefer et al., 1976; Schneider, Osterroht, 1977; Sims et al., 1977; Roots, Peikre, 1978; Perttilä et al., 1982, and others).

There is no single opinion on the relationship between the concentration of residual chlorinated hydrocarbons in the organs and tissues of fish and the level of lipids in them. In the majority of studies authors have discovered a reliable correlation between concentrations of DDT and PCBs and lipid levels (Portmann, 1975a; Schefer et al., 1976; Lukinykh et al., 1977; Goerke et al., 1979; Schneider, 1982; Perttilä et al., 1982, and others), although in other studies no correlation has been noted (Henderson et al., 1971; Addison et al., 1972). Earnst and Benville discovered both a positive and negative correlation between chlorinated hydrocarbon concentration and lipid level in several species of marine fish, this probably being related to the uneven distribution and levels of lipids in fish in relation to sex, age, stage of sexual development, and time of year (Earnst, Benville, 1971).

It has been established that consumption of DDT by fish directly from water increases with temperature in proportion to the consumption of oxygen (Reinert et al., 1974; Murphy, Murphy, 1971) and depends on the salinity of the water (Murphy, 1970).

Stenersen et al. (1977) established a reliable correlation between concentrations of DDT and PCBs in fish from the North Sea caught near the southwestern coast of Norway, which points to the widespread distribution of PCBs. This situation is a source of considerable alarm since PCBs, as research has shown (Fraumensi, 1974; Kimbrough, Linder, 1974), possess carcinogenic properties.

In samples taken from the liver of plaice from the western part of the Barents Sea and analyzed by the author, the coefficient of correlation between the level 54/ of DDT and PCBs turned out to be unreliable, although a positive reliable correlation was discovered between the level of lindane and PCBs (r = 0.60), which may be tied to certain characteristics of lindane manufacture in which PCBs are used as a filler to reduce evaporation and improve insecticidal properties.

In studies on metabolic processes in plaice liver from water near the shore of Medvezhii Island a statistical analysis was performed on the relationship between DDT level and the combined level of its metabolites (DDE + DDD) in fish liver (Fig. 6).

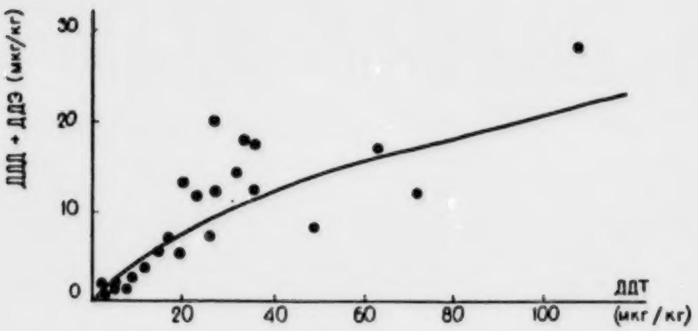

Рис.6. Зависимость между содержанием ДДТ и суммарным содержанием (ДДЭ + ДДД) в печени камбалы-ерша.
$y = 2.567x^{0.614}$, $r = 0.73$.

Figure 6. Relationship between DDT level and combined DDE + DDD level in plaice liver. [Horizontal axis: DDT (μg/kg); vertical axis: DDD + DDE (μg/kg).]

The relationship is approximated by the power function $y = 2.567x^{0.614}$, with a coefficient of correlation equal to 0.73. From Fig. 6 it can be seen that an increase in the level of DDT accumulation in plaice liver was not accompanied by a proportional increase in its metabolites; the ratio between (DDE + DDD) and DDT increased in favor of the latter. This can be explained by that fact that, at low levels of DDT accumulation, its metabolism in plaice liver proceeds more actively. With an increase in the level of DDT accumulation the rate of metabolic processes declines somewhat to a constant value within the range of the residual DDT values examined.

Table 10 presents data on the level of residual chlorinated hydrocarbons in 55/
the organs and tissues of the most abundant species of fish in northern seas.

In analyzing the table data one notes a high variability in the concentrations of chlorinated hydrocarbons in the organs and tissues of fish from various regions. Among the reasons for this variability is the change in pollution levels in the marine environment, which increase steadily in the order: pelagial division – neritic zone – inland seas (Patin, 1979). This tendency is maintained in ichthyofauna as well: the level of DDT and PCBs in marine fish (especially those from inland seas) significantly exceed corresponding values for ocean fish. It is characteristic in the Atlantic Ocean that DDT and PCB concentrations in hydrobionts from the North Atlantic are greater than in related species from the South Atlantic (Harvey et al., 1974).

This is in agreement with a general tendency, noted by S.A. Patin (1977, 1979), toward an increase in pollution levels as one moves from the southern hemisphere to the northern. The accumulation of PCBs primarily in fat-containing organs and tissues is linked to their physico-chemical properties, to a heightened ability to penetrate through cell membranes, a fact confirmed by experimental studies which have made it possible to classify chlorinated hydrocarbons by their ability to accumulate in the following order: PCBs > DDT > lindane (Masek et al., 1970; Hattula, Karlog, 1973). Among the various ecological groups of fish, elevated levels of residual chlorinated hydrocarbons have been discovered in bottom species, which is explained by the accumulation of large quantities of DDT and PCBs in bottom deposits.

The World Health Organization believes that the allowable daily consumption of DDT by the human organism should not exceed 0.3 mg (or 0.005 mg per kg of body weight). This figure has also been taken as a guideline by various countries in setting the maximum permissible concentration of DDT and other organochlorine pesticides in fish and other marine products. The FRG, for example, has enacted a Law on the Maximum Quantity of DDT and Other Pesticides in Food Products of Animal Origin (Gerlach, 1981). According to this law, the concentration of substances of the DDT type in marine fish must not exceed 2 mg/kg (with the exception of the 56/

European eel *Anguilla anguilla*, the salmon *Salmo salar* and the sturgeon *Acipenser sturio*, the flesh of which may contain 3.5 mg/kg). No more than 5 mg/kg of DDT are permitted in fish liver and fish oil.

In the USSR, in accordance with the list "Allowable Residual Quantities of Pesticides in Food Products" and in supplements to it approved by the USSR Minzdrav (Ministry of Public Health) of 24.03.77 (no. 1735-77), 24.08.79 (no. 2052-79) and 21.04.81 (no. 2390-81) residual DDT and lindane up to 0.2 mg/kg are permitted in [fresh] fish and canned fish.

Based on the data of Table 10, it can be said that the fish delivered to northern seaports contain an average DDT level of approximately 0.01 mg/kg. A person who consumes 200 g of fish in a week is ingesting no more than 0.002 mg (0.0003 mg per day) of DDT, which represents one thousandth of the permissible daily norm adopted by the WHO. According to scientists in the FRG (*Ernährungsbericht*, 1976) the DDT concentration in fish is only 1/10 that in meat, sausage and other products prepared from the flesh of domesticated animals.

As regards PCBs, data from the WHO indicates that various skin eruptions can be noted in people having a daily consumption of 0.07 mg of PCBs per kg of body weight (WHO, 1976). No allowable daily consumption [level] has been established by the WHO. In a number of countries, however, restrictions – maximum allowable concentrations of PCBs in fish - are employed. In the USA, for example, the Food and Drug Administration allows PCB concentrations up to 5 mg/kg (Gerlach, 1981), and in France up to 1 mg/kg is allowed (Rentsh, 1982).

In summarizing the available data on PCB levels in fish from northern seas, the following conclusion can be made: the amount of PCBs in fish reaching the consumer is, on average, less than 0.1 mg/kg. This means that a person ingests with fish less than 0.003 mg of PCBs as a daily average, but this constitutes only a small portion of the PCBs [reaching the consumer] from other sources.

The literature contains no data on the level of petroleum hydrocarbons in fish of northern seas, and only T.L. Shchekaturina (1985) has studied the hydrocarbon content in fish of the Barents Sea.

Table 10 57/

Level of residual chlorinated hydrocarbons in abundant species of fish in northern seas (mg/kg of wet wt)

Report area	Type of sample	ΣDDT	PCBs	Source
1	2	3	4	5
Cod *Gadus morhua*				
Baltic Sea				
Coastal waters of Sweden	Muscles	0.063	0.033	Jensen et al., 1969
Coastal waters of Denmark	Liver	11.3 - 22.0	2.4 - 4.9	Westöö, Noren, 1971
Coastal waters of the FRG	Liver	7.5 - 31.7	–	Huschenbeth, 1973
Coastal waters of Sweden	Liver	4.4 - 18.0	1.4 - 9.5	Jensen et al., 1972a
Coastal waters of the FRG	Liver	Up to 60.0	–	Priebe, 1978
Gulf of Finland and Gulf of Bothnia	Muscles	0.43 - 1.4	1.7 - 5.0	Tervo et al., 1980
Coastal waters of Poland	Liver	7.2	8.8	Falandysz et al., 1980
Northern part of the sea	Liver	0.3 - 1.0	2.3 - 5.3	Wideström et al., 1981
Open part of the sea	Liver	0.29	0.005 - 0.2	Kullenberg, 1981
Western part of the sea	Liver	0.59	2.87	Schneider, 1982

(Table 10 continued)

1	2	3	4	5
North Sea				
Coastal waters of England	Muscles	0.012	–	Robinson et al., 1967
Coastal waters of Sweden	Muscles	0.005	0.019	Jensen et al., 1969
Same	Muscles	0.005	0.020	Jensen et al., 1972a
Coastal waters of the FRG	Muscles	30.0	–	Huschenbeth, 1973
Coastal waters of England and Wales				Portmann 1979
winter, 1970	Muscles	0.001 - 0.007	0.007 - 0.31	
	Liver	0.026 - 0.530	0.400 - 12.0	
summer, 1970	Muscles	0.003 - 0.008	0.51 - 0.033	
	Liver	0.46	4.4	
winter, 1971	Muscles	0.006 - 0.014	0.009	
	Liver	0.25 - 1.7	0.5 - 8.3	
summer, 1971	Muscles	0.004 - 0.005	0.012 - 0.025	
	Liver	0.77 - 1.3	2.6 - 12.0	
winter, 1972	Muscles	0.004 - 0.010	0.02 - 0.049	
	Liver	0.42 - 1.7	1.1 - 11.0	
summer, 1972	Muscles	0.003 - 0.006	0.015 - 0.031	
	Liver	0.7 - 2.7	2.8 - 18.0	
winter, 1973	Muscles	0.005 - 0.006	0.020 - 0.027	
	Liver	0.64 - 1.4	2.3 - 4.6	
summer, 1973	Muscles	0.003 - 0.004	0.015 - 0.068	

(Table 10 continued)

1	2	3	4	5
Open part of the sea	Muscles	0.003 - 0.050	–	ICES, 1974
Open part of the sea	Muscles	0.005	–	Ten Berge, Hillebrand, 1974
Coastal waters of Norway, 1974	Liver	0.65 - 11.4	0.78 - 1.5	Kveseth et al., 1979
Coastal waters of England, 1974	Muscles	0.005	0.030	Murray, 1979
	Liver	0.87	5.0	
Open part of the sea	Muscles	0.004	0.01	
	Liver	0.393	3.0	
Coastal waters of England, 1975	Muscles	0.003	0.02	Murray, 1981
	Liver	0.61	8.4	
Coastal waters of the FRG	Muscles	2.1 - 4.9	12.0 - 29.0	Schaefer et al.,1976
	Liver	0.96 - 5.2	8.9 - 22.0	
	Whole	4.4	24.0	
Southern part of the sea, 1974 - 1975	Liver	–	39.0	Kerkhoff et al., 1977
1976 - 1978	Liver	–	48.0	
Northern part of the sea	Liver	–	5.0 - 8.0	
Open part of the sea	Fish fat	0.42	–	Bruggemann et al., 1976
Coastal waters of England	Muscles	0.009	–	ICES, 1977
Open part of the sea	Liver	–	5.1	Freemann, Uthe, 1979

(Table 10 continued)

1	2	3	4	5
Coastal waters of Belgium	Muscles	–	0.005	Vandamme, Baetmann, 1982
Central portion of the sea, 1972	Muscles	0.006 - 0.01	0.03 - 0.07	Ernst et al., 1976
	Liver	0.03 - 1.8	0.3 - 8.1	
North Atlantic				
Northwestern part, 1971 - 1972	Liver	5.2	5.1	Sims et al., 1977
	Fat	1.2	2.5	
Open areas	Muscles	0.011	0.038	Harvey et al., 1974
	Liver	2.7	22.0	
Open areas	Muscles	0.037 - 0.04	–	Huschenbeth, 1973
Northwestern part	Liver	8.1	–	Sims et al., 1975
	Muscles	0.05	–	
	Fat	2.0	–	
St. Lawrence Strait	Liver	1.8	4.1	ICES, 1977
Coastal waters of Greenland	Liver	0.19 - 0.3	0.44 - 0.46	
Northwestern part	Liver	–	5.0 - 8.0	Kerkhoff et al., 1977
Norwegian Sea				
Open part	Liver	10.0	–	Gerlach, 1981
Open part	Liver	12.0 - 40.0	–	Bjerk, 1973

(Table 10 continued) 61/

1	2	3	4	5
Open part	Muscles	0.02	–	ICES, 1977
Kristiansand Fjord	Liver	0.9	6.3	Brevik, 1978
Open part	Liver	0.1 - 14.5	0.7 - 7.5	Brevik et al., 1978
Barents Sea				
Open part	Liver	0.31	1.7	ICES, 1977
	Muscles	0.03	–	
Open part	Liver	0.009	0.03	Murray, 1981
	Muscles	0.003	0.01	
Southeastern part	Liver	0.22	–	Savinova et al., 1981
	Muscles	0.0005	–	
Waters off Medvezhii Island	Muscles	0.0005	0.002	Author's data
	Gonads	0.0016	0.003	
		Ocean perch *Sebastes marinus*		
North Atlantic, Georges Bank	Liver	1.3	1.5	Harvey et al., 1974
	Muscles	0.073	0.190	
Denmark Strait	Muscles	0.032	0.360	
Barents Sea, southwestern part	Liver	0.076	–	Savinova et al., 1981
	Muscles	0.020	–	
Barents Sea, southwestern part	Muscles	0.01 - 0.03	0.04 - 0.09	Author's data

(Table 10 continued) 62/

1	2	3	4	5
Pollack *Pollachius virens*				
Barents Sea, southeastern part	Liver	0.295	–	Savinova et al., 1981
	Muscles	0.017	–	
North Atlantic, Georges Bank	Liver	3.0	45.0	Harvey et al., 1974
	Muscles	0.003	0.037	
Common catfish *Anarhichas lupus*				
Barents Sea, southeastern part	Liver	0.123	–	Savinova et al., 1981
	Muscles	Not detected	–	
Northern catfish *Anarhichas denticulatus*				
Barents Sea, waters off Medvezhii Island	Liver	0.06	0.053	Author's data
	Muscles	0.019	0.021	
	Gonads	0.078	0.080	
Spotted catfish *Anarhichas minor*				
Barents Sea, southeastern part	Liver	0.123	–	Savinova et al., 1981
	Muscles	Not detected		

(Table 10 continued) 63/

1	2	3	4	5
Barents Sea, waters off Medvezhii Island	Liver	0.028	0.007	Author's data
	Muscles	0.0002	Not detected	
Barents Sea, shore of Rybachii Peninsula	Gonads	0.023	0.036	
	Muscles	Not detected	Not detected	
Plaice *Pleuronectes platessa*				
Barents Sea, southeastern part	Liver	0.014	–	Savinova et al., 1981
	Muscles	Not detected	–	
English Channel	Liver	0.12 - 0.5	0.4 - 2.2	Ernst et al., 1976
	Muscles	0.003 - 0.007	0.01 - 0.04	
Skagerrak	Liver	0.16	0.4 - 0.6	Eder et al., 1976
	Muscles	0.008	0.03 - 0.05	
Bullrout *Myoxocephalus scorpius*				
Barents Sea, southeastern part	Liver	0.107	–	Savinova et al., 1981
	Muscles	0.009	–	
Starry ray *Raja radiata*				
Barents Sea, southeastern part	Liver	0.039	–	
	Muscles	Not detected	–	

(Table 10 continued) 64/

1	2	3	4	5
Barents Sea, waters off Rybachii Island	Muscles	0.028	0.029	Author's data
		Plaice *Hippoglossoides platessoides limandoides*		
Barents Sea, southeastern part	Liver	0.147	–	Savinova et al., 1981
	Muscles	0.003	–	
North Sea, central part	Liver	0.05 - 0.1	0.4 - 0.6	Schaefer et al., 1976
	Muscles	0.004 - 0.005	0.03 - 0.05	
Barents Sea, shore of Rybachii Peninsula	Liver	0.008	0.02	Author's data
Barents Sea, waters off Medvezhii Island	Liver	0.058	0.079	
		Haddock *Melanogrammus aeglifinus*		
Barents Sea, southeastern part	Liver	0.037	–	Savinova et al., 1981
	Muscles	0.004		
North Atlantic, Georges Bank	Liver	0.4	8.8	Harvey et al., 1974
	Muscles	0.002	0.030	
Denmark Strait	Muscles	0.003	Not detected	
Barents Sea, waters off Medvezhii Island	Liver	0.02 - 0.03	0.045 - 0.047	Author's data

(Table 10 concluded) 65/

1	2	3	4	5
Barents Sea, shore of Rybachii Peninsula	Liver	0.027	0.059	
	Muscles	0.006	0.008	
	Gonads	0.02	0.001	
Poutassou *Micromestistius poutassou*				
Barents Sea, waters off Medvezhii Island	Muscles	0.006	0.002	
Capelin *Mallotus villosus*				
Barents Sea, Kildinskaya Bank	Whole	0.007	0.009	

NOTE: the dash indicates no determination made.

4.4. Factors affecting accumulation of chlorinated hydrocarbons in fish

66/

Since the phenomenon of accumulation includes the interaction of a substance with an organism, factors which determine the degree of accumulation of any substances should include characteristics of both the substance and the organism.

The author studied the the level of bioaccumulation of organochlorine pesticides (DDT, DDE, DDD and lindane) and PCBs as a function of certain fish characteristics (weight, size, age, coefficient of nourishment[13] level, sex, percentage level of lipids) and of the area of catch, based on representatives of two massively-occurring species of Barents Sea ichthyofauna: cod and plaice.

In order to evaluate the comparative effect of the above-cited factors, the information index of effect[14] was chosen, which, unlike the coefficient of correlation, for example, does not measure the strength of the linear link but characterizes [instead] the relationship between factors in a general way, evaluating their inter-specificity (Kastler, 1960). Information indices of effect are utilized in biological research (Kastler, 1960; Puzachenko, Moshkin, 1969; Beer, 1972) although they have not been used up to now in ecotoxicological research.

With the aid of the information indices an attempt was made to evaluate the comparative effect of various factors on the accumulation of DDT and its metabolite DDE in the liver of cod from the coastal part of the Barents Sea and on the accumulation of DDE, DDD, DDT, γ-HCH (lindane) and PCBs in the liver of plaice caught off the shores of Medvezhii Island. In the first case the following characteristics were taken for analysis: weight, size, coefficient of nourishment, age, sex, area where caught; in the second case – weight, size, coefficient of nourishment, and percentage level of lipids.

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[13] The term 'upitannost' literally means 'fatness' and is used to indicate how well fed an animal is (Tr.).

[14] Russian 'informatsionnyi pokazatel vliyaniya'. The rather literal translation could not be verified. Another possible translation might be 'information influence factor'. (Tr.)

The computed information indices of effect (Aivazyan et al., 1985) are shown in tables 11 and 12. The values of the coefficients are presented in the tables in descending order of intensity of effect.

Table 11

Evaluation of DDT and DDE bioaccumulation in cod liver in relation to weight (W), size (L), age (T), coefficient of nourishment (Q), sex (S) and area where caught (R)

ξ	ζ	$H(\xi)$ bit	$H(\zeta)$ bit	$H(\xi,\zeta)$ bit	$I(\xi,\zeta)$ bit	$C\ \xi/\zeta$
DDT	Q	1.926	1.766	2.979	0.713	0.404
	R	1.926	2.318	3.427	0.817	0.352
	T	1.926	2.009	3.244	0.691	0.344
	L	1.926	1.855	3.306	0.475	0.256
	W	1.926	1.635	3.177	0.384	0.237
	S	1.926	0.983	2.830	0.079	0.080
DDE	R	1.978	2.318	3.459	0.837	0.361
	T	1.978	2.051	3.306	0.688	0.341
	Q	1.978	1.766	3.218	0.526	0.298
	L	1.978	1.841	3.430	0.393	0.213
	W	1.978	1.667	3.302	0.344	0.206
	S	1.978	0.983	2.903	0.058	0.059

As can be seen from Table 11, the greatest effect on the level of DDT accumulation in the liver of cod from coastal waters of the Barents Sea is given 67/ by the coefficient of nourishment, which is due to the known ability of organochlorine pesticides to dissolve in fats.

Of great interest is the fact that the second most important is the ratio between level of DDT accumulation and the area where caught. Based on this, a graph was constructed for meridional distribution of ΣDDT pollution levels in cod liver (Fig. 7). From Fig. 7 it can be seen that the level of ΣDDT accumulation decreases from west to east from the area of the Ainovy Islands to the Guba Orlovka [Orlovka Bay]. In the waters off Kharlov Island, situated further to the east, the level of accumulation of ΣDDT was somewhat higher, which can probably be explained by local pollution of the water near the shore of the island. Numerous colonies of seabirds are found here, and additional contaminants may enter the water along with their excretions.

68/

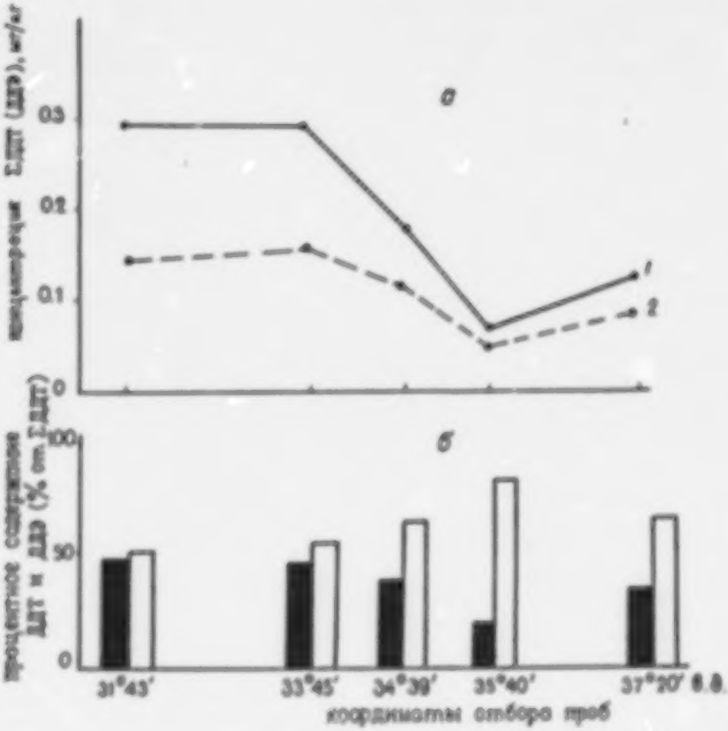

Рис.7. Меридиональное распределение степени загрязненности ДДТ и метаболитом ДДЭ печени трески, отловленной в прибрежье Баренцева моря.

а) концентрация Σ ДДТ (I) и ДДЭ (2); б) процентное соотношение ДДТ:ДДЭ от Σ ДДТ.

Figure 7. Meridional distribution of DDT and DDE metabolite contamination levels in the liver of cod caught in coastal waters of the Barents Sea.

а) concentration of ΣDDT (1) and DDE (2); б) ratio DDT:DDE as a percent of ΣDDT.

[а-vertical axis: ΣDDT (DDE) concentration, mg/kg.

б-horizontal axis: coordinates of sample collection site (degrees east longitude); б-vertical axis: level of DDT and DDE (% of ΣDDT)]

The age of the fish had the next greatest effect on the ΣDDT accumulation level. No correlation was detected, however, between the age and ΣDDT level in cod liver. 69/

Analysis of the effect of the factors examined on DDE accumulation in cod liver indicated that the level of DDE bioaccumulation depends to a greater extent on the area where caught (Fig. 7).

The age of the fish held second place in terms of the intensity of the effect on the DDE bioaccumulation level. This is connected with age-related changes in the activity of the xenobiotic detoxification system in fish of this species. The level of cytochromes P-450 and b_5 and benzpyrenehydroxylase activity were used as the criteria for the organisms' ability to metabolize foreign compounds. Studies conducted with cod of various age groups showed an increase in the level of cytochromes P-450 and b_5 with age (Khokhryakov, 1982). Additional statistical processing of the material determined the existence of a reliable correlation between the age of the cod and the level of accumulation in its liver of DDE.

Among the factors considered, the sex of the fish showed the least effect on the level of accumulation of both DDT and DDE in cod liver (Table 12).

Analysis of information indices for the effect of various factors on the accumulation of chlorinated hydrocarbons in the liver of plaice indicated that lipid content exercised the greatest effect on bioaccumulation levels for all of the contaminants considered. Study of the relationship between the level of chlorinated hydrocarbons and lipid levels in the liver of plaice was based on ΣDDT accumulation (Fig. 8). This relationship is approximated by the function $y = 0.782e^{0.174x}$, with a coefficient of correlation equal to 0.81. The graph of the relationship in a semi-logarithmic coordinate system takes the form of a straight line.

Table 12

Evaluation of chlorinated hydrocarbon bioaccumulation in plaice liver in relation to weight (W), size (L), age (T), percentage lipid level (P), and coefficient of nourishment (Q).

ξ	ζ	$H(\xi)$ bit	$H(\zeta)$ bit	$H(\xi,\zeta)$ bit	$I(\xi,\zeta)$ bit	$C\ \xi/\zeta$
DDD	P	2.310	1.969	3.607	0.671	0.341
	Q	2.310	1.917	3.702	0.552	0.273
	W	2.310	2.284	4.021	0.573	0.251
	T	2.310	2.284	4.073	0.521	0.228
	L	2.310	1.955	3.831	0.434	0.222
DDE	P	1.990	1.969	3.198	0.761	0.386
	Q	1.990	1.915	3.535	0.370	0.193
	T	1.990	2.284	3.857	0.418	0.183
	W	1.990	2.284	3.892	0.382	0.167
	L	1.990	1.952	3.771	0.174	0.089
DDT	P	2.284	1.969	3.319	0.934	0.474
	Q	2.284	1.915	3.607	0.592	0.309
	W	2.284	2.284	3.961	0.608	0.266
	T	2.284	2.284	4.004	0.565	0.247
	L	2.284	1.955	3.814	0.425	0.218
γ-HCH	P	2.254	1.969	3.555	0.670	0.339
	W	2.254	2.284	3.798	0.741	0.324
	Q	2.254	1.915	3.650	0.518	0.270
	T	2.254	2.284	3.978	0.560	0.245
	L	2.254	1.955	3.952	0.257	0.131
PCBs	P	2.294	1.969	3.552	0.711	0.361
	L	2.294	1.955	3.554	0.694	0.355
	W	2.294	2.284	3.814	0.765	0.335
	Q	2.294	1.915	3.788	0.421	0.220
	T	2.294	2.284	4.158	0.420	0.184

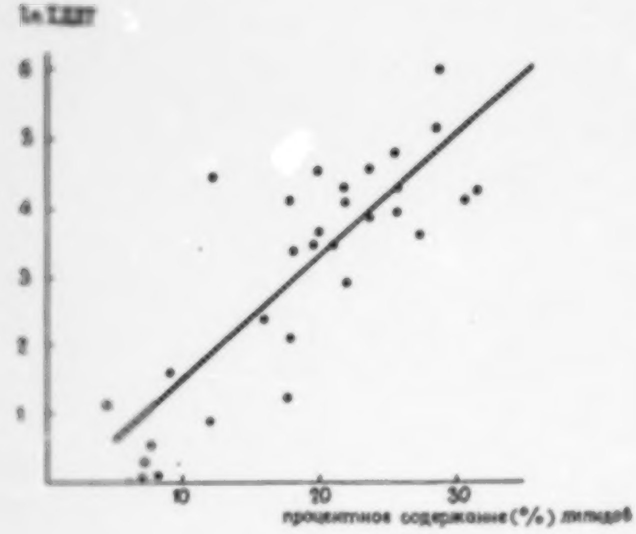

Рис.8. График зависимости между содержанием липидов в печени камбалы-ерша и концентрацией ΣДДТ в этом органе.
$y = 0.782\ e^{0.174x}$, $r = 0.81$.

Figure 8. Relationship between lipid content and ΣDDT concentration in plaice liver. [Horizontal axis: lipid content (%); vertical axis: ln ΣDDT.]

For organochlorine pesticides of the DDT family and lindane the factors affecting their accumulation levels in plaice liver, in terms of degree of effect, occur in the same order, with the exception of DDE, the accumulation of which is most affected by age. This fact confirms the conclusions made earlier 71/ regarding the relationship between the level of DDE accumulation and age of the cod. One may presuppose that such a tendency is also characteristic for certain other species of fish and is due to the biochemical processes of detoxification which result in the formation of DDE.

DDT is metabolized into DDD in fish by intestinal microflora (Greer, Paim, 1968; Cherrington et al., 1969, and others), with the result that the age of the fish exercises no substantial influence on the level of this metabolite in plaice liver.

For PCBs the greatest effect, after lipid content, on its accumulation in plaice 72/
liver came from the size and weight characteristics of the fish. This leads to the conclusion about the comparatively greater influence of adsorption processes on accumulation of PCBs rather than of organochlorine pesticides. In last place in terms of strength of effect among all of the cited factors affecting PCB accumulation in plaice liver is age of the fish. This fact is evidence for the absence of chronic PCB contamination in the report area.

Thus, the results of the analysis performed made it possible to identify the factors having the greatest effect on chlorinated hydrocarbon levels in the liver of marine fish and demonstrated the effectiveness of utilizing information indices of effect in toxicological research.

Chapter 5. EFFECT OF CONTAMINANTS ON PHYTOPLANKTON OF NORTHERN SEAS

5.1. Effect of chlorinated hydrocarbons on phytoplankton

The available literature contains evidence that phytoplankton is highly sensitive to the effect of chlorinated hydrocarbons. This sensitivity manifests itself in changes in rate of growth and cell division in algae and in their morphological degradation and destruction (Morgan, 1972; Glooschenko, Glooschenko, 1977; Picer et al., 1979; Kleppel, McLauglin, 1980; Fernandes et al., 1983). DNA synthesis is suppressed in algal cells (Lal Rup, Saxena, 1980), pigment levels decline and chloroplast lamellae are distorted (Glooschenko, Glooschenko, 1977), and the overall amino acid concentrations decrease (Crezuga, Gierasimov, 1977), contributing to changes in the intensity of photosynthesis.

The toxic resistance of neritic and oceanic forms of phytoplankton is 73/
unequal, due to differences in levels of water pollution and related adaptation of algae, and to species diversity and successional changes in phytocoenoses in the course of anthropogenic impact (Menzell et al., 1970; Fischer et al, 1973; Shulyakovskii, 1980).

Chlorinated hydrocarbons present a special risk for phytoplankton of northern seas since an increase in the volume of oil produced on the shelf and its possible leakage intensifies the toxic effects of DDT, its analogs, and of PCBs. This is because DDT and PCBs dissolved in oil penetrate the cell membrane more readily (Patin, 1979).

Toxicological studies of marine phytoplankton are comparatively few in number and are devoted primarily to studying the effect of toxicants on mixed and pure cultures of algae (Wurster, 1968; Menzel et al., 1970; Portmann, 1972; Cox, 1972; Mosser et al, 1974; Walsh et al., 1977; Harding, Phillips, 1978a; Kleppel, McLaughlin, 1980, and others), and this complicates the use of the results obtained to evaluate real situations arising in nature. For this reason it would be preferable to carry out such studies on natural communities of phytoplankton under conditions that approximate natural ones as closely as possible. Up to now *in situ* experiments have examined the effect of chemical toxicants on primary production of the Baltic Sea (Patin, 1979; Kaitala, Maximov, 1982; Kaitala et al, 1983), of certain areas of the North Atlantic (Moore, Harris, 1972; Thomas et al., 1977; Shulyakovskii, 1980), and of the Barents Sea (Savinova et al., 1984; Savinova, Savinov, 1987).

Study of the effect of organochlorine pesticides on primary production of the Barents Sea centered on DDT and kepone. Phytoplankton specimens were collected in an open bay of the Eastern Murman in the incoming tide. The timing of the experiment involving DDT (early May, 1983) coincided with the period of mass blooms of the golden alga *Phaeocystis sp.* Of the kepone-containing phytoplankton samples collected for the experiment in the middle of May, 1983, *Chaetoceros* sp., which accounted for 70% of the sample, and *Fragilaria oceanica*, which accounted for 30%, predominated. The time of exposure for the DDT experiment was 4 and 12 h. In the experiment with 74/ kepone the sample exposure time was 4, 8 and 12 h. At each of these points in the experiment samplers[15] were brought to the surface, one after another, and a

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[15] The Russian '*sklyanki*' means 'phial' or 'bottle'. Here it is assumed to be some sort of plankton sampler or collector bucket. (Tr.)

200 ml sample collected from each to determine the primary production of phytoplankton and of chlorophyll "a", after which the samplers were again lowered to the exposure horizon.

Results of the experiment to study the effect of DDT on primary production of phytoplankton are given in Figure 9a, b. A comparison of the two graphs permits the following conclusions:

an increase in DDT concentration to 100 µg/l led to a sharp increase in primary production – to almost 300% as compared with controls – at the 4-hour mark;

an increase in a toxicant's exposure time reduced somewhat the degree of stimulation of the photosynthesis process in the presence of a DDT concentration of 100 µg/l (up to 180% as compared with controls), while the photosynthesis maximum (220% compared with controls) began to decline at a DDT concentration equal to 10 µg/l. The highest concentration of chlorophyll "a" after a 12-h exposure (170% compared with controls) was recorded in samplers with a DDT concentration equal to 100 µg/l, which is the result of a certain delay in the increase of the chlorophyll "a" concentration as the primary production level increased;

a 12-h exposure to DDT at a concentration of 1000 µg/l has an observable inhibitory effect both in terms of primary production and of chlorophyll "a".

75/

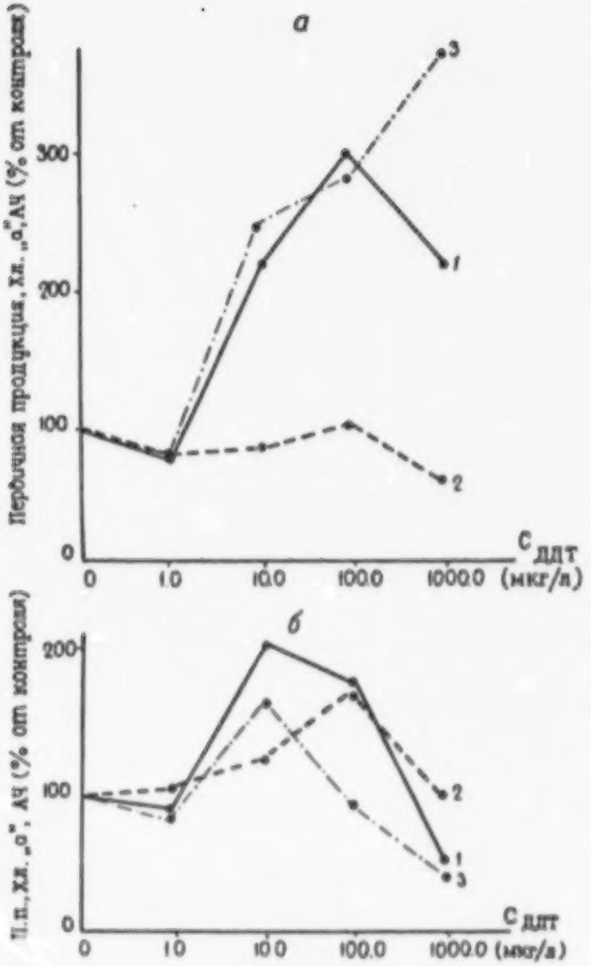

Рис.9. Влияние ДДТ на первичную продукцию (I), содержание хлорофилла "а" (2) и ассимиляционное число (3) природного фитопланктона Баренцева моря.

а – экспозиция 4 ч; б – экспозиция I2 ч.

Figure 9. Effect of DDT on primary production (1), level of chlorophyll "a" (2) and the assimilation number (3) of natural phytoplankton of the Barents Sea.

а - 4 h exposure; б - 12 h exposure.

[а & б, horizontal axis: concentration of DDT (μg/l); vertical axis: primary production, chlorophyll "a", assimilation number (% of controls)]

From the results of the determination of primary production and chlorophyll "a", values were computed for the assimilation number (AN) – one of the indices for the physiological activity of phytoplankton. The term 'assimilation number' was proposed by the German physiologists Willstätter and Stoll (1918). As defined by them, the AN is the maximum quantity of carbon dioxide that can be restored per unit time by a unit quantity of chlorophyll in cells or tissue under optimal conditions. In the present-day literature dealing with phytoplankton ecology, AN is taken to mean a ratio between the intensity of photosynthesis and chlorophyll "a" (Strickland, 1960; Finenko, 1970). The experiment 76 / demonstrated that an increase in the exposure time to DDT leads to a sharp decrease in the AN value – in other words, to a decrease in the physiological activity of the phytoplankton.

The growth, over time, in the inhibitory effect of DDT can be confirmed by the data, previously obtained by the author, on the effect of this pollutant on primary production of phytoplankton in experiments based on a similar methodology (with a 24-h exposure) in waters off the shore of Severnyi ostrov [Severnyi Island (Tr.)], Novaya Zemlya, and the open part of the Barents Sea in 1981. In these experiments the manifestation of the inhibitory effect relative to photosynthesis at DDT concentrations as low as 10 - 12 μg/l was a characteristic feature of the response to DDT in the level of primary production of phytoplankton collected both in coastal waters and in the open sea. In these experiments, some stimulation of photosynthesis remained only at a DDT concentration of 1 μg/l[16].

According to Patin (1979), there exist at least two independent reasons for the temporary increase in the intensity of alga vitality in a toxic medium. One of these is related to the general sequence, well-known in toxicology, of phases in the development of toxicosis when poisoning is preceded by stimulation of an organism's vital activity and activation of physiological-biochemical regulatory mechanisms. There exists yet another possible cause for this phenomenon: destruction of the metabolic links in the system phytoplankton - bacteria. A toxic addition of the DDT and PCB type disrupts interrelationships between phytoplankton and bacteria in the former's contact zone. This may happen, for example, by chemical bonding of the metabolites or as the result of significant differences in the toxic resistance of bacterial microflora and of the phytoplankton, when, at specific levels of toxicity, photosynthesizing cells may temporarily enjoy preponderantly favorable conditions for development.

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[16] An alternative reading of this last phrase might be "some stimulation of photosynthesis remained at a DDT concentration of as little as 1 μg/l" (Tr.).

The experiment with kepone indicated that, by comparison with DDT, this contaminant is more toxic for phytoplankton at the same concentrations as DDT. The maximum value for primary production obtained in the experiment and measured in samplers with a kepone concentration of 100 μg/l (following 8-h 77/ exposure) was 150% of controls (Fig. 10. With an increase in the exposure time to 12 h the level of primary production at this concentration fell to 40% of controls. The maximum value for primary production shifted toward a reduced concentration of toxicant (10 μg/l) and stood at control levels.

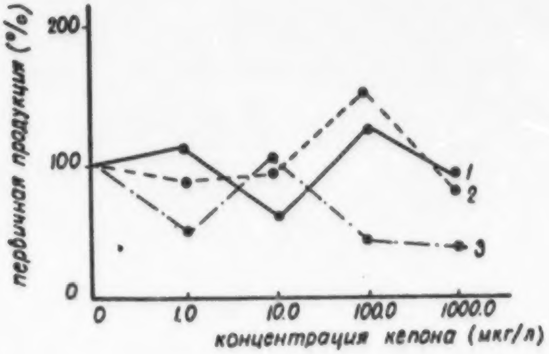

Рис.I0. Влияние кепона на первичную продукцию природного фитопланктона Баренцева моря. Время экспозиции: I - 4 ч; 2 - 8 ч; 3 - I2 ч.

Figure 10. Effect of kepone on primary production of natural phytoplankton of the Barents Sea. Time of exposure: 1 - 4 h; 2 - 8 h; 3 - 12 h.

[Horizontal axis: kepone concentration (μg/l); vertical axis: primary production (%).]

The majority of ecotoxicological experiments have focused on examining the effects of individual toxicants on biological systems. Under real conditions, however, several pollutants are usually present simultaneously. For this reason, only by confirming that the effects of two or more toxins on biological systems when found together are not interrelated would it be possible to evaluate the consequences produced by these contaminants when each is studied separately in the laboratory.

Experimental work on the joint or combined harmful effect of several contaminants on any biological system can be easily presented as special cases of a multiple factor experiment (Maksimov, 1977).

Short-term *in situ* experiments on the effects of kepone and p,p'-DDT on primary production of natural phytoplankton in the Barents Sea were performed 78/ on the basis of a 2^2 full factorial experiment (Savinova, Savinov, 1987). In order to clarify the effects of various additions of organochlorine compounds on primary production, six two factor experiments were set up in which the entire range of concentrations selected for study were broken down into three intervals for p,p'-DDT (1, 100, 1000 μg/l) and two intervals for kepone (1 and 100 μg/l). The toxins used were acetone solutions of p,p'-DDT and kepone prepared in such a manner that the pre-selected toxicant concentrations were established as an aliquot of the solutions was introduced into the production samplers. Samples of natural phytoplankton taken in June 1984 in the southeast part of the Barents Sea served as the material for the experiments. The phytoplankton samples primarily contained *Sceletonema costatum* (60%), *Chaetoceros* sp. (20%), and *Nitzschia seriata* (15%).

From the results of the experiments regression coefficients were computed and their statistical significance was verified. The resulting regression equations for six experiments are given below in the order of their coefficient values.

$$y_I = 20.2 - 1.8x_1 + 1.2x_2 + 4.1x_1x_2 , \qquad (1)$$

$$y_{II} = 13.7 - 4.8x_1 + 3.8x_2 - 1.6\,x_1x_2 , \qquad (2)$$

$$y_{III} = 8.2 + 3.7x_2 + 1.5x_1x_2 , \qquad (3)$$

$$y_{IV} = 11.4 + 1.3x_1 - 10.1x_2 - 1.0x_1x_2 \qquad (4)$$

$$y_V = 9.3 - 3.4x_1 - 8.2x_2 + 3.0x_1x_2 , \qquad (5)$$

$$y_{VI} = 6.3 - 5.6x_2 . \qquad (6)$$

where x_1 - p,p'-DDT;

x_2 - kepone.

Analysis of the regression equations and their related particular effects permits the following conclusions:

photosynthesis is suppressed in the presence of one of the toxicants (p,p'-DDT or kepone) in a concentration of 1 μg/l; under their combined influence at concentrations up to 1 μg/l a positive effect results (experiment I);

an increase in the concentration of one of the toxicants to 100 μg/l 79/
produces a negative effect; as the concentration of the second toxicant increases to 1 μg/l (with concentration of the first at 100 μg/l) the particular effect of the second toxicant decreases noticeably and remains positive (experiment II, IV);

the absence of the regression coefficient for x_1 in equation (3) – due to its insignificance – shows that, as the concentration of p,p'-DDT increases from 100 to 1000 μg/l, its effect on primary production in the experiment was determined by the combined action with kepone (experiment III);

in experiment V, under the combined effect of p,p'-DDT and kepone in concentrations from 1 to 100 μg/l, the negative effect of each of these toxicants was somewhat weaker as the level of the other increased;

the absence from equation (6) of coefficients of x_1 and x_1x_2 indicates that 80/
p,p'-DDT in concentrations from 100 to 1000 μg/l, while remaining an inhibitory factor, did not exercise a substantial effect on primary production; a decrease in production during the experiment depended to a much greater extent on the level of kepone concentration.

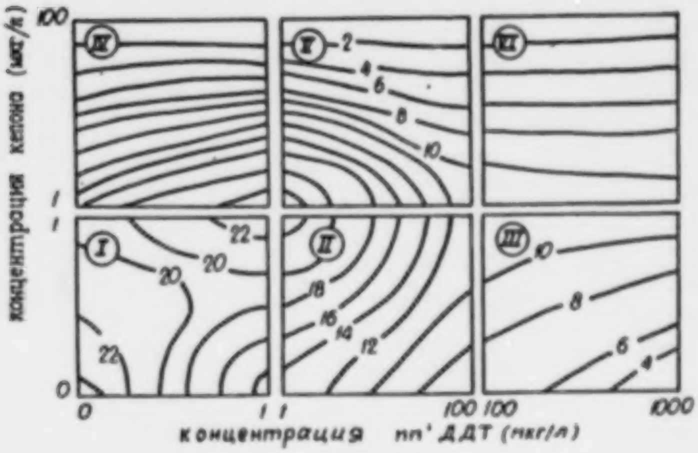

Рис.II. Комбинированное действие п,п′-ДДТ и кепона на первичную продукцию фитопланктона (проекции поверхностей отклика на факторную плоскость).

Римские цифры – номера экспериментов; арабские цифры – уровни первичной продукции, мгС/м^3.

Figure 11. Combined effect of p,p'-DDT and kepone on primary production of phytoplankton (response surfaces projected onto a factor plane[17]).
Roman numerals – experiment numbers; Arabic numerals – primary production levels (mgC/m^3).
[Horizontal axis: p,p'-DDT concentration (μg/l); vertical axis: kepone concentration (μg/l).]

Figure 11 shows combined response projections with primary production equal-value levels superimposed. Of greatest interest in the research on these toxicants' effect on photosynthesis under real ocean conditions are the results of the first experiment inasmuch as the contamination concentrations selected for this experiment differ only slightly from those recorded in natural bodies of waters. It can be seen from Fig. 11 that as addition levels of p,p'-DDT and kepone increased from 0 to 1 μg/l, the character of their combined effect on primary production changes qualitatively – suppression is replaced by a certain stimulation of photosynthesis. Further increases in toxicant concentrations lead to a decline in primary production levels.

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[17] This translation of the Russian 'faktornaya ploskost' could not be confirmed (Tr.).

Thus, it is possible to conclude that an interdependence exists between p,p'-DDT and kepone in their combined effect on primary production, especially at concentration intervals that may exist in natural bodies of waters.

Heavy metals, together with organochlorine compounds, belong to a group of contaminants which are the most widely distributed in the hydrosphere and the most threatening to the viability of aquatic systems. The mechanism whereby heavy metals exercise their toxic effect on marine phytoplankton is rather fully reflected in the review of More and Morel-Lourens (1983).

Experimental works have studied the effect of heavy metals on phytoplankton populations (Wong et al., 1980; Seisuma et al. (1984), on the intensity of photosynthesis, on respiration and hydrocarbon levels, and on the synthesis of protein and chlorophyll (Patin, 1979; Rai et al., 1981; Egorov et al., 1984). But the presence of several contaminants in the marine environment requires study of the combined effect of toxicants of different kinds. In this context there have been a limited number of works studying the combined effect 81/ on phytoplankton of various metals (Patin, 1979; Seisuma et al., 1984), of metals and PCBs (Kaitala, Maximov, 1982; Kaitala et al., 1983), of metals and detergents (Patin, 1979), and of DDT and drilling fluid components (Savinov, Bobrov, 1988).

In connection with the development of oil production on the shelf of the Barents Sea there is interest in experimental study on the combined effect on natural Barents Sea phytoplankton communities of drilling fluid components entering the sea during drilling and of DDT type compounds which have been present in the sea water for decades. Such studies have been conducted in the form of a series of two-factor experiments (Savinov, Bobrov, 1988). There has been study on the combined effect of DDT and 'ferumchlorolignosulfonate' ('FCLS') – the most toxic water soluble component of drilling fluids. Toxicant concentrations were determined at intervals of 0 - 40 mg/l 'FCLS' and 0 - 100 µg/l DDT. The results of each experiment served as the basis for constructing surfaces of response of primary production to the combined effect of 'FCLS' and DDT and for projecting them onto a factor plane (Fig. 12).

As seen in Fig. 12, in the absence of the second contaminant the level of primary production decreased over the entire interval of 'FCLS' concentrations, with the minimum (51% of controls) occurring at an 'FCLS' concentration of 1 mg/l. The combined effect of 'FCLS' and DDT was more complex:

> under the combined effect of 'FCLS' and DDT in concentrations of 0 - 1 mg/l and 1 - 100 µg/l, respectively, the inhibitory effect occurring in the presence of one of the contaminants and absence of the other declined somewhat; an increase in the concentration of 'FCLS' from 1 to 10 mg/l led to an increase in primary production from 60 to 150% of controls, while the effect of DDT in this case was insignificant;
>
> under the combined effect of 'FCLS' at concentrations of 1 - 10 mg/l and DDT at 0 - 1 µg/l, inhibition and stimulation of photosynthesis alternated; the level of primary production increased to 180%, and reached 300 % of controls at a 'FCLS' concentration of 40 mg/l – in which case the increase in primary production level was primarily dependent on the DDT concentration;
>
> a further increase in DDT concentration from 1 to 100 µg/l, with a 82/
> simultaneous increase in 'FCLS' from 10 to 40 mg/l, led to a decline in primary production to 110% of controls.

Results of the experiments indicated that, upon the entry of drilling fluids into a marine environment containing DDT, the processes of forming new organic substances may be somewhat stimulated, and these processes may be suppressed in the absence of DDT.

The effect of contaminants on primary production of phytoplankton in the Norwegian Sea has gone virtually unstudied. This was the goal in a series of four experiments carried out by the author to study the combined effect of organochlorine pesticides and heavy metals on the vitality of natural phytoplankton communities in the Norwegian Sea. Table 13 shows the range of concentrations and design of the factor experiments.

After a statistical analysis of the experiments' results, regression coefficients and their level of significance were computed. In this way the following regression coefficients were obtained:

$$y_{I} = 5.4 + 1.2x_1x_2\,. \tag{7}$$

$$y_{II} = 6.0 + 1.8x_1x_2\,. \tag{8}$$

$$y_{III} = 2.0 + 0.7x_1x_2\,. \tag{9}$$

$$y_{IV} = 1.0 - 0.4x_1 - 0.4x_2\,. \tag{10}$$

The presence of non-zero coefficients of x_1x_2 in regression equations (7) - (9) demonstrates the interdependence of factors x_1 and x_2. In each of these three experiments, an increase in the level of any one of the factors being examined brought about a change in the character of the effect of the second – from negative to positive – resulting in a certain stimulation of photosynthesis as compared with controls. Thus, levels of primary phytoplankton production in the Norwegian Sea in the presence of the toxicants examined in these experiments depended basically on the combined effect of the latter on the photosynthesis process.

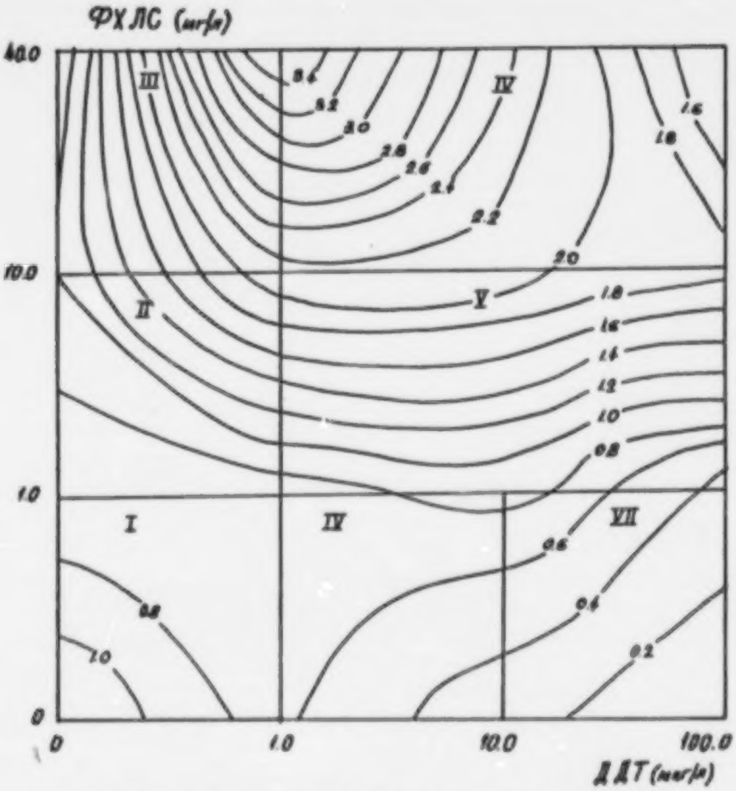

Рис.12. Комбинированное действие ФХЛС и ДДТ на первичную продукцию фитопланктона Баренцева моря (проекции поверхностей отклика на факторную плоскость).

Римские цифры - номера экспериментов; арабские цифры - уровни первичной продукции, мгС/м³.

Figure 12. Combined effect of 'FCLS' and DDT on primary production of phytoplankton in the Barents Sea (projections of response surfaces onto a factor plane).
Roman numerals – experiment number; Arabic numbers – primary production levels (mgC/m^3).
[Horizontal axis: DDT (µg/l); vertical axis: 'FCLS' (mg/l).]

Results of the experimental work in the Barents and Norwegian seas permit the following conclusions: the effect of the organochlorine compounds studied (DDT, p,p'-DDT, kepone) on primary production of phytoplankton in northern seas is not single-valued, for the following reasons: 83/

possibility of both inhibiting and stimulating effects on photosynthesis; 84/
change in the nature of the effect due to interaction of different contaminants, such as two organochlorine compounds (p,p'-DDT and kepone) or the presence of heavy metals (DDT-copper, DDT-lead) and drilling fluid components (DDT-'FCLS').

Table 13

Design of 2^2 factorial experiments for combined effect of DDT and copper and DDT and lead on primary production of natural phytoplankton communities in the Norwegian Sea

2^2 factorial experiment number	Replication number	x_1	x_2	y
		DDT concentration (μg/l)	concentration of copper (I-II), lead (III-IV) (μg/l)	primary production (% of controls)
I	1	0	0	100
	2	0.5	0	61
	3	0	10.0	69
	4	0.5	10.0	106
II	1	0	0	100
	2	3.0	0	45
	3	0	10.0	73
	4	3.0	10.0	119
III	1	0	0	100
	2	0.5	0	30
	3	0	100.0	73
	4	0.5	100.0	118
IV	1	0	0	100
	2	3.0	0	52
	3	0	100.0	62
	4	3.0	100.0	10

Any of the above-mentioned factors represent an undoubted risk to primary production in northern seas. But the heterogeneity of phytoplankton species and the diversity of its metabolic characteristics prevent the entire phytoplankton coenosis in a polluted body of water from being disrupted. The principal factor 85/ in this stability is the systemic buffering effect (Kamshilov, 1973) due to differences in the rates of consumption by individuals, to populations of different sizes, and to uneven distribution both of the toxic substance in space and between individual components of the coenosis. Phytoplankton stability as a whole depends on numerous factors, which according to L.P. Braginskii et al. (1987) include:

the buffering and detoxifying function of aquatic microflora, capable of breaking down many organic contaminants and converting them into harmless, easily assimilatable compounds;

seasonal dynamics of phytoplankton, whereby forms constantly change and domination by one group of algae passes to another, with the result that intoxicated groups of algae are constantly being replaced by other, uncontaminated species;

the high rate of reproduction of algae;

the tendency for certain algae, at a definite stage of vegetation, to form aggregates, to coalesce and form complex algo-bacterial assemblages capable of withstanding the most potent contaminants.

Thus, experiments in the laboratory with single-species cultures or *in situ* in model systems give a picture only of the potential risks of intoxification, that is, they do not take into account the above-mentioned factors which, together with hydrologic and physico-chemical factors, can counter the effect of a toxicant and affect its fate in the water. Nevertheless, as regards toxicants already thoroughly polluting the hydrosphere (especially in cases of chronic contamination), such studies are important and essential by providing predictive data.

5.2 Effect of petroleum hydrocarbons on phytoplankton

The biogenic nature of certain petroleum hydrocarbons is apparently responsible for the fact that these compounds have a weakly expressed toxicity for hydrobionts. According to Tsvylev and Tkachenko (1985) the toxicity of petroleum hydrocarbons is 2 - 3 orders [of magnitude] lower than that of other 86/ contaminants of global distribution. Research conducted by English scientists indicates that in the period from 1948 to 1982 the population structure of North Sea phytoplankton underwent no substantial changes (Raid, 1987). In a study on the effects of effluents from drilling platforms in the North Sea, no significant negative effect was noted on phytoplankton communities. Only an indirect polluting effect was observed, which was linked to a reduction in copepod population leading to an increase in the phytoplankton population (Gamble et al., 1987). Similar data was obtained in the Dutch sector of the North Sea (Scholten, Kulper, 1987).

A negative effect from a large concentration of petroleum hydrocarbons is linked with the lipophilic nature of petroleum hydrocarbons and of their emulsifiers and detergents. Because of this property, the toxicants easily penetrate the lipoprotein cell barriers of the algae, provoking certain metabolic and morphological disruptions. The larger molecules of polycyclic aromatic petroleum hydrocarbons penetrate slowly into the cell and part the lipid layer, whereas low-boiling fractions penetrate quickly and dissolve in the membrane lipids, altering the spacing between structural elements, thus causing the lipid layer to swell and the cell membrane to burst (Biggs et al., 1979).

A characteristic reaction of algae to petroleum hydrocarbons is, as in the case of chlorinated hydrocarbons, a stimulation of photosynthesis at low concentrations and a suppression at high concentrations. The stimulating effect of oil may be related both to the phase nature of phytoplankton's reaction to a toxicant as well as to the presence in the oil of growth regulators and trace quantities of metals which perform the role of microelements (Hsiao, 1978; Hsiao et al., 1978).

The variability of phytoplankton's response may be influenced by such factors as the type of oil (its chemical composition), the means of dispersal, temperature, duration of exposure, the sensitivity of particular alga species, and many others.

Under experimental conditions, no inhibitory effect was noted for oil concentrations of less than 1 mg/l on development of a monoculture of various algae species (Prouse et al., 1976; Parsons et al., 1976). In field studies in the 87/ vicinity of oil spills in the North Sea no increase in phytoplankton production was recorded, while it was observed that petroleum hydrocarbons shift the structure of single-cell alga communities toward a predominance of small forms. Petroleum hydrocarbon contamination in a concentration of 40 - 80 μg/l leads to intensive development of nannoplankton forms having a diameter of approximately 10 μm, with a predominance of flagellated forms (Parsons et al., 1976). Small forms dominated in the phytoplankton communities following the leakage of oil near the wreck of the tanker "Torrey Canyon" (Smith, 1968), a phenomenon linked to suppression of large cells with greater sensitivity to oil and to stimulation of life processes of the more stable small forms. The same pattern was confirmed also in experimental work with mixed cultures of marine single-cell algae (Shulyakovskii, 1980). A change in species composition and cell size is the most sensitive indicator of functional disruption in phytoplankton due to the effect of oil pollution. These changes have been recorded at oil product concentrations from 0.05 to 5.0 mg/l (Patin, 1979).

The most sensitive to the effect of oil are the diatoms. O.G. Mironov (1970) established experimentally that diatoms perish within 24 h at a liquid fuel concentration of 100 μg/l or less. In response to one-percent oil extracts of the kind spilled during the "Torrey Canyon" catastrophe, the growth rate of diatoms decreased by 10% (Lacare, 1969). Experiments provide evidence for a negative effect from [crude] oil and solar oil on the development of small flagellated algae at concentrations of 0.001, 0.01 and 0.1 ml/l (Mironov, 1985).

Research on the species-specific sensitivity of algae has revealed that the toxicity of fuel oil is related to the presence of poorly soluble high [molecular weight] aromatic fractions rather than of paraffins and asphaltenes (Batterton et al., 1978). Results from studies by Norwegian scientists on the toxic effect of the water-soluble fraction of oil from the "Ekofisk" field in the North Sea have shown that toxicity is primarily the result of the low-polar oxidized compounds, including peroxides, ketones, sulphoxides and others (Ostgaard et al., 1987). 88/

There is evidence that the degree of toxicity of oil depends on light. A 20% concentration of Kuwaiti oil reduces the growth of diatoms (Lacaze, Villidon-de-Naide, 1976). Intensification of oil's toxicity in response to light is linked to the transformation and formation of toxic compounds (Ostgaard et al., 1984; Sydnes et al., 1985). Interesting studies have been carried out by Norwegian scientists (Sydnes et al., 1985) which show that a change in solar radiation is of major significance to changes in oil toxicity in the Arctic. During the Arctic spring and summer, when solar radiation is at its maximum, the toxicity of oil increases significantly due to the formation of photooxidation products. This phenomenon may represent a serious risk to the survival of phytoplankton in this region.

Besides light, other factors, such as concentrations of biogenic elements, also have an impact on algae's sensitivity to oil (Karydis, 1981).

Analysis of the literature on the effect of petroleum hydrocarbons on phytoplankton of the northern seas permits the following conclusions:

in the open sea and in unpolluted shelf zones with background levels of petroleum products, oil does not exercise a substantial effect on phytoplankton communities:

existing concentrations of oil in anthropogenically altered coastal regions and estuaries and at oil spill sites appears to give rise primarily to changes in the overall population, biomass and production, and also alter the structure of the community toward a predominance of small forms;

in Arctic regions with specific light conditions, evaluation of toxic effect must take solar radiation into consideration.

Aquatic ecosystems possess a definite potential for resisting the influence of 89/ chlorinated and petroleum hydrocarbons by mobilizing homeostatic mechanisms developed in the course of evolution and elaborated in response to anthropogenic influences. But since residual quantities of these and other persistent toxicants will continue to circulate in the aquatic ecosystems of northern seas for many years to come, the problem of biological consequences due to the presence of these substances in the aquatic environment remains a current concern and requires additional study.

Chapter 6. CHEMICAL POLLUTION AND SEABIRDS

6.1 Oil pollution and seabirds

A large number of seabirds become the victims of oil pollution, but more birds perish every year from chronic pollution than die in any single catastrophic spill. Scientists estimate that between 150 000 and 450 000 seabirds die every year merely from the effects of oil pollution in the North Sea and North Atlantic (Tanis, Mörzer Bruijns, 1968; Bourne, 1976). Lemmetyinen (1966) estimates annual losses in certain areas of the Baltic at 10 000 to 40 000; approximately 11 000 seabirds perish each year along the coast of Holland (Tanis, Mörzer Bruijns, 1968); and 25 000 to 50 000 near the coast of Great 90/ Britain (Bourne, 1976). On the coast of Belgium and Holland in the period 1958 - 1962 an average of 12 birds suffering from oil pollution were counted each year along 1 km of shoreline, and in the period from 1963 to 1968 this number had increased to 47 birds (Parslow, 1971). These figures, however, do not reflect the real picture of pollution since the corpses reaching the shore represent, according to various estimates, only a portion of the total losses at sea. It has been suggested that the number includes only 5 - 15% of total losses (Nelson-Smit, 1977).

Even the comparatively small oil spill (700 t) from the "Hamilton Trader" tanker accident near North Wales resulted in the death of 6 000 to 10 000 seabirds (Bourne, 1976). Mortality rates following a larger spill can surpass this figure by a factor of 10 - 100. For example, the collision of two tankers near Chatham, Massachusetts, reduced the winter population of common eider from 500 000 to 150 000 (Nelson-Smit, 1977). The spill from the "Torrey Canyon" killed between 40 000 and 100 000 birds (primarily razorbills) on both coasts of the English Channel. The razorbill population was reduced to one-ninth of its former size (Bourne, 1976). The list of losses of seabirds due to catastrophic oil spills can be continued *ad libidum.* Special studies having been devoted to this question, and monographs, revues, and materials from ornithological symposiums have been published.

The risk of oil pollution to birds has increased due to the development in recent decades of oil production on the ocean shelf. According to Swedish ornithologists, over 100 000 swimming birds have perished over the last 20 years off the coast of Sweden as a result of oil leaks from production operations on the shelf and from shipping (Oil ..., 1982). Another source of serious concern among scientists is the massive (and so far still uncalculated) deaths of birds in collisions with drilling platforms and ships (Matishov, 1989).

In studies of the toxicity of oil for seabirds it has been established that swallowing the oil leads to development of lipoid pneumonia, severe irritation of the intestines, fatty liver, and an increase [in the size of] the adrenal gland. 91/
Disorders of the nervous system and various necroses have also been observed (Hartung, Hunt, 1966; Beer, 1968) The toxic effects of oil depend on a number of factors, especially on the age of the birds. Cessation of growth has been noted in the young of herring gulls that swallow crude oil (Gorman, Simms, 1978). Oil can affect embryos through the egg shell. Experiments carried out under both field and laboratory conditions (White et al., 1979) have demonstrated that significantly fewer birds hatched from heron[18], tern and gull eggs kept in oil concentrations of 0.5 - 20 mg/l, as compared with controls. For

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[18] The Russian 'tsapli' include bitterns and egrets as well as herons (Tr.).

herons, 61% of the embryos perished, and 56% and 85% for terns and gulls. In some nature sanctuaries in the 1950's liquid fuel was sprayed on eggs to control the population of gulls and cormorants (Gross, 1951). Following the "Torrey Canyon" tanker accident, according to Rittinghaus (1956), terns contaminated with oil had abnormally low numbers of offspring; the same phenomenon was evidenced in studies on contamination of ducks following the same accident (O'Connor, 1967). In experimental work Hartung (1965) demonstrated that a single dose of 2 g of lubricating oil per 1 kg of weight swallowed by birds rapidly halts egg-laying and suppresses reproduction.

Contamination of birds with oil causes feathers to stick together, with the result that the air layer insulating the body is destroyed. This is an especially serious risk for Arctic regions where the thermal insulating layer that protects the body from chilling is a critical survival factor.

A thin layer of oil penetrates and mats the feathers, even a rather small quantity of oil being sufficient to damage the feathers. A number of authors (Hawkes, 1961; Bourne, 1976) believe that even a small spot of oil on a bird's breast is enough to cause it to die in time from hypothermia, especially in northern seas.

As the result of an oil spill in 1982 off the northeastern shores of the Magdalen Islands (Canada) 1500 seabirds died. An autopsy indicated that they had died not from swallowing the oil but from damage to their feathers (Oil ..., 1982).

According to Gerlach (1981), in the winter of 1980 - 81 more than 60 000 92/
birds died from oil pollution in the Skagerrak. In 1983 the nesting sites of thousands of seabirds were destroyed on the shores of Helgoland, Scotland,

Norway and the Kola Peninsula. In the North Sea alone oil pollution in the winter of 1983 was the cause of death of over 80 000 seabirds (Müller, 1983).

Birds' specific habits mean the possibility of contamination through contact with floating slicks. Divers and razorbills sit low in the water, and so oil may cover them completely. When diving into the water, and without being aware of it, they may enter a layer of oil that covers their body. Observations carried out by Bourne (1968) indicate that the birds' first reaction upon encountering oil (as in cases of greatest danger) is to dive, which leads to them being even more contaminated. Gulls usually leave quickly a site of heavy contamination. As studies in the Baltic Sea have shown, long-tailed ducks prefer to land on oil slicks, possibly because wave action is always reduced on such a site or possibly because of the slick's resemblance to a school of fish (Lemmetyinen, 1966).

Norwegian ornithologists (Norderhaug, 1976; Norderhaug et al., 1977) divide the birds of the North Atlantic and Barents Sea that suffer the most from oil pollution of these waters into four groups: alcids, ducks, diving birds, gulls and gull-like birds. The most affected in Norwegian waters are the black guillemot, Brünnich's murre, puffin, razorbill (2-3 species), diving birds (2-3 species), the common cormorant, the shag, and the pochard (11 - 12 species), as well as various gulls[19] (especially the lesser black-backed gull, herring gull and kittiwake). As oil exploration begins on the shelf of the Barents Sea Norwegian scientists are finding there is a lack of information on the ecology of the birds of this region, especially as regards problems connected with drilling.

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[19] The Russian term 'chaika' (pl. 'chaiki'), commonly translated as 'gull' or 'sea gull', is a broader term than its English counterpart and seems to more closely correspond to the order 'Charadriiformes'. At one point in the original it is referred to as an 'order'. In the present work it has been translated simply 'gull'. (Tr.)

Research has been conducted in Norway in recent decades on the dynamics of seabird populations and migration routes. There have been studies on the level of bioaccumulation of heavy metals and chlorinated hydrocarbons in the bird organism which intensify oil's toxicity. The need for comprehensive research on pollution levels of chlorinated and petroleum hydrocarbons in birds is evident inasmuch as these substances intensify each 93/
other's toxicity. Moreover, birds contaminated with oil are more subject to contamination from lipophilic chlorinated hydrocarbons. According to English scientists, nearly 10 000 seabirds (primarily murres[20]) were washed ashore in the Irish Sea in the winter of 1969-1970. Only a few specimens were heavily affected by oil, the overwhelming majority of them having perished from high concentrations of PCBs in their bodies (Bourne, Mead, 1969).

6.2 Chlorinated hydrocarbons and seabirds

Although chlorinated hydrocarbons were detected in significant quantities in the organs and tissues of fish-eating birds in England in the early 60's (Moore, Tatton, 1965), this did not attract much attention until the "Seabird Group" reported the death of many thousands of murres and razorbills on the shores of the Irish Sea in 1969-1970 (Bourne, Mead, 1969). Studies revealed that high concentrations of chlorinated hydrocarbons in the organs and tissues of fish-eating birds and raptors were the reason for a decline in their numbers, for disruptions in their population structure, and, in some cases, their death: in the USA (Cottam, Bolen, 1973; Wiemeyer et al., 1978; Young et al., 1979); the

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[20] The Russian 'kaira' broadly includes *Uria* sp. (Tr.).

Netherlands (Koeman, 1971, 1972b, 1975); FRG (Bernd, 1978); Norway (Bjerk, Holt, 1971; Bourne, 1976); Belgium (Joiris et al., 1979); Denmark (Franzmann, 1982); France (Keck et al., 1982); Canada (Vermeer, Reynolds, 1970; Gilbertson, 1975; Vermeer, Peakall, 1977); Sweden (Jensen et al., 1970; Olsson et al., 1973; Renberg et al., 1978); Finland (Särkkä et al., 1978; Soikkeli, 1979); and in many other countries.

A theory exists that organochlorine pesticides of the DDT type are at the origin of mutations in ocean viruses transmitted through the "zooplankton - fish" [food] chain to seabirds, which in their subsequent migrations around the world spread a flu virus responsible for pandemics, as happened with the "Hongkong-2" virus (Soloukhin, 1974).

Accumulation of chlorinated hydrocarbons in the organs and tissues of fish-eating birds has been studied rather well. Table 14 presents some data on 94/
levels of residual ΣDDT and PCBs in seagulls[21].

Levels of chlorinated hydrocarbon pollution in seabirds vary very widely and depend on many factors, such as the species of bird and its ecology – migration routes, type of food, sex, age, quantity and composition of lipids, season, and so forth (Keith, 1970; Bourne, 1976; Anderson, Hickey, 1976).

Sexual differences in the accumulation of DDT and PCBs in birds is due to their loss in females in the course of non-specific mobilization of fats during egg-laying and nest-sitting. Finnish scientists (Särkkä et al., 1978) noted that DDT levels are significantly greater in male seagulls than in females. Work done to study contaminant levels in terns in southwest Finland revealed the same pattern (Lemmetyinen, Rantamäki, 1980). Differences in the degree of chlorinated hydrocarbon accumulation between sexes in raptors and fish-eating birds may also be due to differences in the effectiveness of detoxification processes (Lemmetyinen et al., 1982).

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[21] Here and elsewhere the Russian 'morskie chaiki' - literally 'sea gulls' - has been so translated, although some sources give 'great black-backed gull' as a translation. The reader is requested to keep this alternative meaning in mind. (Tr.)

Research conducted in the field has determined that roughly identical quantities of DDE can be found in the adult female and in the eggs it lays (Bogan, Newton, 1977). It has been shown experimentally that the penetration of pesticides and PCBs inside the egg depends on the porosity and thickness of the shell (Danielle, Lutz-Ostertag, 1976), with PCBs penetrating into eggs more easily than pesticides of the DDT family (Lemmetyinen, Rantamäki, 1980). In seabirds, chlorinated hydrocarbons accumulate principally in the yolk (Fry, Toone, 1981).

Research on the degree of contamination of seabird organs and tissues due to residual DDT and PCBs has shown that maximum contaminant levels are recorded in hatchlings and decline with increasing age. This pattern has been observed in terns, shags, and several species of gulls (Robinson et al., 1967; Jensen et al., 1970; Charnetski, 1976).

Table 14

ΣDDT and PCB levels in organs and tissues of gulls (mg/kg of wet weight)

Locality	Type of sample	Number of samples	PCB	ΣDDT	Source
1	2	3	4	5	6
			Rissa tridactyla		
North Sea coast 1963	Eggs	6	Not determined	(0 - 0.7)	Moore, Tatton, 1965
1964	Eggs	6	Not determ.	(0.4 - 1.2)	
North Sea, coast of Great Britain, 1965	Eggs	26	Not determ.	0.25+ (0.21 - 0.29)	Robinson et al., 1967
Pacific Ocean, coast of USA, 1966	Muscles	–	Not determ.	1.3	Risebrough et al., 1967
Davis Strait, 1971-1975	Liver	1	3.2	0.13+	Bourne, 1976
	Muscles	1	1.9	0.08+	
Medvezhii Island, 1971 - 1975	Liver	1	1.6	0.08+	
	Muscles	1	3.7	0.18+	
Coast of Northern Scotland, 1971 - 1975	Liver	9	5.2 (0.3 - 20.7)	0.17+ (0.04 - 0.5)	
	Muscles	9	2.96 (0.4 - 5.0)	0.08+ (0.01 - 0.14)	

(Continuation of table 14)

1	2	3	4	5	6
Northeast coast of Great Britain, 1971 - 1975	Liver	2	2.1 5.5	0.26 0.55	Bourne, 1976
	Muscles	2	2.2 3.0	0.25 0.30	
Northwest coast of Great Britain, 1971 - 1975	Liver	1	505.0++	23.0++	
	Muscles	1	61.0++	28.0++	
Larus fuscus					
Lake Päijänne, Finland, 1972 - 1974	Liver	47	8.0 (0.84 - 47.83)	9.04 (1.54 - 34.41)	Särkkä et al., 1978
	Muscles	44	6.71 (0.27 - 18.87)	5.27 (0.08 - 16.83)	
Larus argentatus					
Baltic Sea, coast of Sweden, 1968 - 1970	Muscles	4	18.0	Not detected	Olsson et al., 1973
Lake Päijänne, Finland, 1972 - 1974	Liver	13	11.27 (0.77 - 29.48)	7.50 (0.21 - 20.83)	Särkkä et al., 1978
	Muscles	16	13.49 (0.68 - 37.71)	8.26 (1.04 - 22.05)	

1	2	3	4	5	6
Baltic Sea, coast of Poland (young birds) 1975 - 1976	Liver	4	4.8++ (2.4 - 8.7)	3.9++ (1.9 - 6.0)	Falandysz, 1980
	Muscles	4	6.6++ (3.2 - 12.0)	6.5++ (2.8 - 13.0)	
	Fat	3	63.0++ (29.0 - 100.0)	71.0++ (39.0 - 130.0)	
Baltic Sea, coast of Poland (adult birds) 1975 - 1976	Muscles	3	100.0++ (23.0 - 150.0)	75.0++ (13.0 - 140.0)	Falandysz, 1980
	Liver	4	280.0++ (140.0 - 530.0)	180.0++ (50.0 - 320.0)	
Baltic Sea, coast of Finland, 1978	Eggs	20	18.1	7.0	Lemmetyinen et al., 1982
	Muscles				
	of hatchlings	13	33.0	17.2	
	of chicks (2 - 4 weeks old)	9	2.1	0.8	
	of young gulls	20	(1.7 - 2.2)	(1.0 - 1.1)	
	of adult gulls	25	(18.7 - 22.1)	(6.7 - 6.9)	
North Atlantic, coast of Canada, 1968 - 1969	Eggs	7	30.0	13.0	Vermeer, Reynolds, 1970
Same location, 1971 - 1973	Eggs	43	4.1	2.0	Zitko et al., 1972

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1	2	3	4	5	6
St. Lawrence Strait, 1968 - 1972	Eggs	25	16.0	6.4	Vermeer, Peakall, 1977
North Atlantic, coast of USA	Eggs	30	7.76 (0 - 32.0)	2.13 (0.34 - 12.6)	Szaro et al., 1979
Faeroe Islands, 1975	Muscles Liver	1 1	13.4+ 12.6+	3.15+ 3.35+	Bourne, 1976
East coast of Scotland, 1971 - 1975	Muscles Liver	5 5	0.64 0.52	0.22 0.24	
Coast of Denmark, 1980	Eggs Brain	50 8	Not determined Not determined	1.40+ 1.90+	
Coast of Norway, 1967	Eggs	8	4.2	1.5	
Coast of Norway, 1976	Eggs	203	8.49	1.57+	
Lake Ontario	Eggs	58	138.0 (116.6 - 157.0)	17.56 (14.8 - 21.2)	Norstrom et al., 1978

NOTE: dash indicates no data; + - DDE only ; ++ - birds found dead.

Research on the seasonal dynamics of DDT and PCB accumulation in adult and young herring gulls revealed that a gradual accumulation of chlorinated hydrocarbons is observed in young gulls after fledging, after which a dynamic 99/ equilibrium sets in that is partially connected with seasonal accumulation of fat reserves. The maximum level of contaminants is reached in both age groups during the winter decline in fat reserves prior to the breeding season. Subsequently, DDT and PCB levels were restored to equilibrium level (Anderson, Hickey, 1976).

Bogan and Newton (1977) demonstrated that there is an increase in DDE levels in birds' internal organs as fat levels in the body decrease. Thus, when the body's lipid level drops below 1.5%, the DDE level in a bird's brain increases quite sharply.

Birds' ecological characteristics are of great significance for the accumulation of chlorinated hydrocarbons. Differences in DDT and PCB accumulation levels are frequently due to feeding conditions (Henry et al., 1977; Gaskin, Holdrinet, 1978; Conrad, 1979). It has been shown that fish-eating birds contain higher concentrations of chlorinated hydrocarbons than land birds (Koivusaari et al., 1976; Oelke, Russel, 1980; Falandysz, Szefer, 1982). The birds' seasonal migrations play no small role in contaminant accumulation (Lindvall, Low, 1979; Lemmetyinen et al., 1982; White et al., 1983). The results of field and experimental studies have established that lethal concentrations of DDT and PCBs are several tens to several hundreds of mg/kg in the brain of a bird (Bourne, 1976; Stickel et al, 1984). Even small doses of DDT and PCBs in the bird organism, however, provoke changes in a number of morphophysiological and biochemical indicators: levels of hemoglobin, urea, K^+ ions, and erythrocytes; a reduction in alanineaminotransferase activity; a rise in lactate-dehydrogenase and creatinephosphokinase activity in the blood; disruption in the hormone balance due to a reduction in the [size of?] endocrine gland ducts and atrophy of the thyroid gland (Jofferies, Parslow, 1976; Rappe, 1979; Demaret, 1979; Fourie, Hatting, 1979; Buteiko, 1980).

Residual amounts of DDE, by disrupting Ca^{2+} metabolism, cause eggshell thinning in various species of birds (Blus et al., 1972; Cooke, 1973; Anderson et al., 1975; Peakall, 1975; Parker, 1976; Cooke et al., 1976; White et al., 1983).

It has been established experimentally that introducing 2 - 100 mg/kg of 100/

DDT into the yoke sac is accompanied by a feminization of embryos during subsequent incubation. The most pronounced feminizing effect is displayed by o,p'-DDT (Fry, Toone, 1981). Feminization may be one of the possible causes of disruption in bird population structures in areas of high DDT content.

Under the effect of chlorinated hydrocarbons there occur durable and irreversible disruptions of reproductive processes due to suppression of 'carbonilanhydrase' and ATPase activity in the membranes of the oviducts (Peakall, 1975). Disruption of the reproductive function of fish-eating birds under the influence of DDT and PCBs caused the destruction of populations of the brown pelican and a number of other bird species in the USA (Anderson et al., 1969, 1975; Wiemeyer et al., 1978). The greatest mortality among birds whose organs and tissues contain high concentrations of residual DDT and PCBs is observed during the brooding period, apparently as a result of the fact that the female loses up to 40% of her weight in this period and the concentration of the accumulated contaminants in her body increases (Franzmann, 1982).

That hatchlings posses a high degree of sensitivity to the effects of chlorinated hydrocarbons has been confirmed by research carried out on the North Atlantic coast where high mortality among hatched chicks and significant anomalies in embryo development linked to the effects of DDT and PCBs were detected (Hays, Risebrough, 1972).

The problems of the impact of contaminants on birds have been discussed at international ornithological congresses, at the World Conference of the International Council for Bird Preservation (1966), at sessions of the council in Seattle (1975), Aberdeen (1977), Cape Town (1979), 'Utoksler', Saint John's and Hawaii (1982) (Bourne, 1983).

At the 28th conference of the International Council for Bird Preservation, which celebrated its 60th anniversary in 1982, the question was raised once again of a final ban on use of DDT and dieldrin, which have a fatal impact on birds. The need was underscored for an intensification of environmental measures to guard birds against the effects of chlorinated hydrocarbons, and it 101/ was noted that seabirds are an important indicator of the state of the natural environment inasmuch as they clearly depend on changing conditions in marine biocoenoses (Edmond-Blank, 1982).

6.3. Chlorinated hydrocarbons in seagulls of the Murman coastal region

Just how great a role birds do play in biocoenoses of the high latitudes one can judge simply from the fact that this group of vertebrates frequently constitutes here the basis of biomass for the animal population as a whole. Studies to clarify the influence of colonial birds on the biological productivity of coastal regions in the Barents Sea (Golovkin, 1963; Golovkin, Pozdnyakova, 1965) have shown the following: even comparatively small colonies of birds exercise a substantial influence on the level of biogenic elements in near-shore sections of the sea by forming areas with an elevated level of biogenic elements, a prerequisite for creation in coastal areas of local areas with elevated bioproductivity.

The role of birds in the biological balance of northern seas is significant. Thus, for example, in the Barents Sea in the period 1936-1937 birds consumed 10% of the annual catch of fish – 630 000 t (Rass, 1948). But according to S.I. Uspenskii (1969), despite destruction of large quantities of fish stocks, there are no grounds for considering the overwhelming majority of seabirds harmful to commercial fishing. Their feeding focuses, as a rule, on less valuable, non-commercial species of fish and on the refuse from fishing. And all species of Barents Sea gulls – the most numerous order[22] of birds on the Murman coast – are among the "sanitation workers" of the sea (Belopolskii, 1957).

The biology of gulls on the Murman coast has been studied rather completely (Modestov, 1939; Kaftanovskii, Modestov, 1941; Belopolskii, 1957; Gerasimova, 1965). On the Murman coast the most numerous are the black-legged kittiwakes, *Rissa tridactyla L.*, while the second most numerous position belongs to the herring gull, *Larus argentatus* Pont., and to the great black-backed gull, *Larus marinus* L.

Estimates place the total number of kittiwakes[23] in the Arctic and Subarctic is at least 200 000 - 250 000 individuals (Uspenskii, 1969). The largest colonies 102/ of kittiwake are found on the coast of the Eastern Murman, and of the herring gull and great black-backed gull on the Ainovy Islands – 1500 and 1600 pairs, respectively (Gerasimova, 1965).

In order to determine the degree of bioaccumulation of residual chlorinated hydrocarbons in organs and tissues of Barents Sea gulls, great black-backed and herring gulls were shot in May 1979 on the Ainovy Islands, and then kittiwakes in July of the same year in the area of the Guba Podpakhty (Eastern Murman). The chest muscle, liver, brain, heart and spleen of the birds were removed for analysis.

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[22] See earlier footnote on translation of the Russian 'chaika' - 'gull' (Tr.).

[23] Here and elsewhere the term 'moevka' - 'kittiwake' - is used without further qualification. It isn't clear whether this always refers to the species *Rissa tridactyla L.* (Tr.)

Results of gas chromatographic analysis indicated the presence in all samples of DDT, its metabolites, and PCBs in concentrations reaching tens of mg/kg of wet weight. To identify polychlorinated biphenyls, samples were analyzed on a chromatomass-spectrometer OWA-20 (Finnigan). The reconstructed ion chromatograms and mass-chromatograms and their comparison with library data made it possible to identify the composition of PCBs detected in the gull organs and tissues, including Aroclor 1254 (product of the USA). Table 15 gives data on levels of residual chlorinated hydrocarbons in the organism of the gull species examined.

As can be seen from Table 15, the maximum level of chlorinated hydrocarbon accumulation was noted for a majority of seagulls. The average ΣDDT concentration (range of concentration variation given in brackets) in the spleen, heart, chest muscle, liver and brain was 12.931 (0.397 - 62.280), 5.377 (0.530 - 12.420), 3.280 (0.550 - 8.530), 2.273 (0.232 - 6.317), 1.423 (0.0962 - 3.953) mg/kg of wet weight. The ranking in ΣDDT content in the great black-backed gull was spleen → heart → muscles → liver → brain. The average level of residual PCBs was greatest in the heart of the great black-backed gull – 3.95 (0 - 12.1) mg/kg – and somewhat less in the spleen – (3.77 mg/kg). But the maximum PCB concentrations recorded in the gull species examined were detected in the spleen of the great black-backed gull – 16.36 mg/kg. The average level of residual PCBs in the great black-backed gull decreased in the order heart → spleen → liver → muscles → brain.

Table 15

ΣDDT and PCB levels in organs and tissues of birds (mg/kg of wet weight)

Type of sample	No. of samples	p,p'-DDT	o,p'-DDT	p,p'-DDE	o,p'-DDE	p,p'-DDD	ΣDDT	PCBs
1	2	3	4	5	6	7	8	9
				Great black-backed gull				
Muscles	6	0.834 0.102 - 1.779	0.453 0.061 - 1.195	0.489 0.180 - 0.998	0.105 0 - 0.396	1.395 0.133 - 4.461	3.277 0.551 - 8.53	1.72 0.52 - 2.3
Liver	8	0.596 0.058 - 1.760	0.329 0.033 - 1.180	0.376 0.087 - 0.707	0.061 0 - 0.350	0.808 0.054 - 2.550	2.273 0.232 - 6.317	2.48 0.3 - 8.0
Brain	7	0.347 0.002 - 0.908	0.178 0.014 - 0.640	0.372 0.029 - 1.098	0.043 0 - 0.205	0.517 0.025 - 1.098	1.423 0.096 - 3.953	0.83 0.01 - 2.1
Heart	7	2.846 0.053 - 14.10	0.582 0.030 - 1.740	0.521 0.080 - 1.877	0.062 0 - 0.149	1.452 0.057 - 5.059	5.377 0.530 - 12.42	3.95 0 - 12.1
Spleen	5	3.413 0.099 - 16.36	2.238 0.045 - 10.93	2.660 0.106 - 12.74	0.463 0 - 2.290	4.164 0.124 - 19.97	12.931 0.379 - 62.28	3.77 0.3 - 16.36
				Kittiwake				
Muscles	10	0.210 0.129 - 0.314	0.116 0.070 - 0.196	0.090 0.037 - 0.160	0.021 0 - 0.057	0.396 0.200 - 0.823	0.832 0.440 - 1.430	0.992 0.59 - 1.22
Liver	10	0.260 0.078 - 0.475	0.118 0.058 - 0.232	0.263 0.059 - 0.655	0.010 0 - 0.012	0.377 0.107 - 0.909	1.030 0.352 - 1.799	1.128 0.68 - 2.17
Brain	10	0.060 0.023 - 0.109	0.041 0 - 0.109	0.027 0 - 0.088	-	0.099 0.029 - 0.211	0.228 0.092 - 0.453	0.35 0.12 - 0.92
Heart	10	0.274 0.108 - 0.491	0.116 0.048 - 0.212	0.084 0.036 - 0.152	0.040 0 - 0.106	0.436 0.179 - 0.710	0.930 0.395 - 1.466	0.996 0.39 - 1.53

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1	2	3	4	5	6	7	8	9
				Herring gull				
Muscles	6	0.767 0.056 - 1.429	0.790 0.024 - 0.570	0.639 0.070 - 1.306	0.090 0.007 - 0.145	1.140 0.078 - 2.356	3.026 0.235 - 5.179	2.990 0.63 - 3.9[87]
Liver	6	0.826 0.201 - 2.352	0.379 0.074 - 0.937	0.985 0.294 - 2.943	0.083 0 - 0.197	1.198 0.294 - 2.943	3.472 0.883 - 9.372	3.753 1.02 - 8.97
Brain	6	0.325 0.044 - 0.971	0.169 0.019 - 0.388	0.342 0.044 [-] 1.117	0.055 0.003 - 0.217	0.451 0.063 - 1.068	1.343 0.173 - 3.762	1.62 0.30 - 3.95
Heart	6	0.475 0.041 - 1.245	0.368 0.021 - 1.245	0.394 0.005 - 0.834	0.069 0.003 - 0.258	0.757 0.057 - 2.495	2.063 0.130 - 5.870	1.13 0.01 - 2.85
spleen	6	0.705 0.022 - 1.501	0.510 0.016 - 1.501	0.638 0.022 - 1.251	0.079 0 - 0.247	1.004 0.032 - 3.002	2.920 0.092 - 7.250	1.48 0.03 - 3.02

No o,p'-DDD was recorded in a single sample of gull organs and tissues. 105/ The concentration of p,p'-DDD was rather high in all organs and tissues; the average level of p,p'-DDT varied from 32.8% in the spleen to 36% in the brain ([as a percentage] of ΣDDT); the level of o,p'-DDE varied from 1.1 in the spleen to 2.6% in the muscles, and that of p,p'-DDE from 22.2% in the heart to 27.7% in the brain.

As compared with the great black-backed gull, the level of chlorinated hydrocarbon bioaccumulation in the herring gull from this same region (Ainovy Islands) is somewhat lower. The average ΣDDT level in the herring gull decreased in the order liver → muscles → spleen → heart → brain (3.472, 3.026, 2.920, 2.063, 1.343 mg/kg, respectively). The level of p,p'-DDD in the organs and tissues of the herring gull, as in the great black-backed gull, was rather high and varied from 32 (in the spleen) to 36% (of ΣDDT) in the brain.

The level of o,p'-DDE varied from 2.2% (liver) to 3.4% in the heart; p,p'-DDE – from 19.9 (heart) to 27% (liver). The level of residual PCBs in the herring gull decreased in the order liver → muscles → brain → spleen → heart.

Kittiwakes were the least contaminated by residual pesticides of the DDT and PCB group (Eastern Murman). The average concentration of ΣDDT (in mg/kg) in the liver, heart, muscles and brain of kittiwakes was 1.03, 0.930, 0.832 and 0.228, respectively, i.e. 3 - 5 times less than in the organs and tissues of the great black-backed gull.

No o,p'-DDD was recorded in the organs and tissues of the kittiwakes either. The average percentage of p,p'-DDD in the organs and tissues of the kittiwake was somewhat higher than in the great black-backed gull and herring gull. The level of o,p'-DDE varied from 0.4 in the brain to 3.8% in the heart; p,p'-DDE – from 9.5 in the heart to 25% in the liver. Characteristically, the concentration of p,p'-DDE in the organs and tissues of the kittiwake was somewhat lower than in the great black-backed gull and herring gull. The concentrations of residual PCBs detected in the kittiwake organism were minimal for the gull species examined. For all three species of gull no substantial difference was found in metabolite levels of the various organs and tissues.

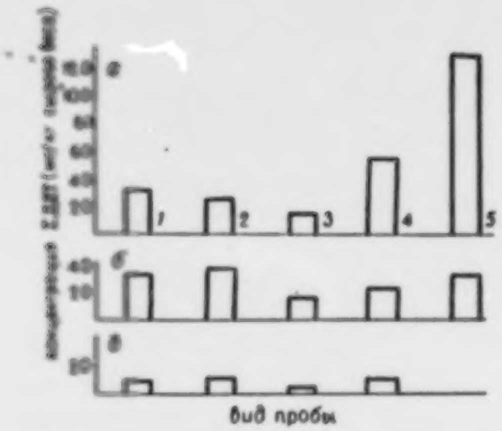

Рис. 13. Сравнительное распределение остаточных количеств ΣДДТ в органах и тканях большой морской (а), серебристой (б) чаек и моевки (в).

1 – мышцы; 2 – печень; 3 – мозг; 4 – сердце; 5 – селезенка.

Figure 13. Comparative distribution of residual ΣDDT in organs and tissues of great black-backed gull (а), herring gull (б) and kittiwake (в).
1 – muscles; 2 – liver; 3 – brain; 4 – heart; 5 – spleen.
[Horizontal axis: Type of sample; vertical axis: ΣDDT concentration (mg/kg of wet weight)

In terms of level of residual PCB and DDT contamination, the gulls fall in the 106/ following order: great black-backed gull → herring gull → kittiwake (Fig. 13), which is related to certain ecological characteristics of the species studied.

The gull species examined can be distinguished both in migration areas as well as in feeding biotopes and methods of obtaining food. Based on an analysis of gull feeding habits in the Murman coastal region, T.D. Gerasimova (1965) divided them into two groups: active ichthyophages, to which the kittiwakes belong, and passive ones (herring gull and great black-backed gull). Fish constitute between 70.5 and 94.5% of the diet of the kittiwake, while the great black-backed gull and herring gull consume basically sick or dead fish

and fishing scraps. Fish caught by them represent only 7.2 - 7.5% of their diet. The littoral of the Ainovy Islands forms the feeding biotope of the great black-backed gull and herring gull. Taking up residence near colonies of eider, the gulls of these species engage in predation. Clearly, the kittiwake's lower level of chlorinated hydrocarbon bioaccumulation can be explained first and foremost by the uniform and less contaminated contents of its diet as compared with 107/
other gull species. The birds' migration areas are also of no small importance in the accumulation of pollutants. Banding has shown (Dementev, 1947) that the kittiwakes winter in western Iceland, Greenland and Newfoundland, where the overall pollution levels in hydrobionts are lower than on the western shores of England and France – migration sites for the great black-backed gull and herring gull.

The ΣDDT level in the liver of the great black-backed gull from the Ainovy Islands is close to that in this same species of gull from lakes in Finland (Särkkä et al, 1978), but the level of ΣDDT accumulation in the chest muscle of Barents Sea gulls is approximately three-fold higher. It should be noted that the liver and muscles of the gull species studied from the Ainovy Islands contained significantly smaller (by a factor of ten) concentrations of PCBs as compared with Scottish gulls. While field and experimental studies have demonstrated that the basic pathway by which chlorinated hydrocarbons enter the body of fish-eating birds is the trophic (Risebrough et al., 1967; Falandysz, Szefer, 1982), differences in PCB and ΣDDT accumulation levels are due to the birds' feeding conditions. According to L.O. Belopolskii (1957) and Gerasimova (1965), the great black-backed gull feeds primarily on marine food sources, which constitute 97.2% of its total diet. The comparatively low overall level of residual DDT in the great black-backed gull from the Ainovy Islands confirms the lower level of this toxin in Barents Sea ichthyofauna as compared with that of the North Sea. The residual PCBs present, identified as components of Aroclor 1254, were apparently accumulated by the gulls during their lengthy winter migrations on the shores of England and France, whose industries utilize this product.

The nature of the distribution of residual chlorinated hydrocarbons in the organs and tissues of the great black-backed gull may be due to re-distribution and a general reduction in lipids during nesting, egg-laying and brooding, at the end of which the gulls were taken for analysis.

The overall accumulation level for residual chlorinated hydrocarbons in organs and tissues of the herring gull from the Ainovy Islands is lower than that for birds of this species nesting on lakes in Finland (Särkkä et al., 1978; Lemmetyinen et al., 1982) and is close to that (only in ΣDDT level) in gulls of 108/ this species from the Faeroe Islands, which, however, contain residual PCBs in higher concentrations than do Barents Sea gulls (Bourne, 1976).

A lethal level of accumulated ΣDDT and PCBs in adult herring gull, according to Falandysz (1980) is 100 mg/kg in the muscles and several hundred mg/kg in the liver. Just such high average concentrations were found in the muscles and liver of dead adult birds of this species on the coast of Poland. A significant level of accumulation in chlorinated hydrocarbons in gulls of the Baltic Sea led to a decline in their numbers and to disruption in the structure of this species' population. Residual chlorinated hydrocarbons in the Barents Sea herring gull were 70 - 100 times below critical levels.

The average PCB level in the muscles and liver of the kittiwake from the shores of Eastern Murman (Guba Podpakhta) did not exceed 1.1 mg/kg, while the level of this toxin in the muscles of kittiwake caught in the Davis Strait, on

Medvezhii Island and on the coast of Northern Scotland was 3 - 5 times higher (Bourne, 1976). In the muscles and liver of the kittiwake from these regions only DDE was recorded in concentrations also exceeding the average metabolite level in kittiwake from the shores of the Eastern Murman.

The uneven distribution of contaminants in the organs and tissues of the kittiwake and the fact that the heart followed the liver – the depository organ – as the site of greatest PCB and ΣDDT accumulation can be explained by the time at which the samples were taken for analysis – July 27, the period when young were being fed. During this period the fat level decreases in the gull's body and the relative level of chlorinated hydrocarbons in internal organs increases (Bogan, Newton, 1977).

Thus, the Barents Sea gulls, in comparison with gulls of this species from other northern sea areas, contain rather small concentrations of residual chlorinated hydrocarbons. But among the analyzed animals of the Barents Sea, the bioaccumulation level of chlorinated hydrocarbons was at a maximum.

In August 1989, chlorinated hydrocarbon levels were studied in herring gulls 109/ caught in the Guba Yarnyshnoi (Eastern Murman). As compared with 1979 (Table 15) the level of residual ΣDDT in the liver and muscles of the herring gull had declined 3-fold over the ten-year period, while PCB levels were virtually unchanged. This reflects the real picture of pollution in the Barents Sea coastal region and agrees with the general tendency of chlorinated hydrocarbon dynamics in marine ecosystems.

At the present time there is no data concerning changes over many years in the numbers and population structure of Barents Sea seagulls that would permit evaluating the negative effects of chlorinated hydrocarbons on the final links in trophic chains and on the ecosystem as a whole. And although the effects of

DDT and PCBs might, theoretically, not become apparent for a long time in systems where the highest consumers are "long-lived" (such as the herring gull, with its life span of 40 - 50 years), the entire range of anthropogenic factors (exploration and drilling on the shelf, shipping, overfishing, etc.) could contribute to a transformation of the ecosystem in a shorter time frame.

CONCLUSION

Research in recent decades has shown just how widely distributed are the zones of chemical pollution in northern seas. As of the present time there have been studies on petroleum and chlorinated hydrocarbon pollution levels in waters and hydrobionts of the North Atlantic and of the North, Baltic, Norwegian and Barents seas, and experiments to determine the effect of toxins on marine organisms of this region.

It has been discovered that residual quantities of PCBs, α- and γ-HCH, DDT and its metabolites and of petroleum hydrocarbons are present everywhere at all water levels in northern seas. Of particular concern is the fact that maximum concentrations of contaminants have been recorded in the most productive regions: in shelf and estuary zones, the euphotic zone, areas of upwellings, the neritic province, and at surfaces of discontinuity between media.

Both individual toxic substances and their combinations disrupt normal functioning of ecosystems in northern seas, which manifests itself in a change in the level of production of organic matter and in a decline in fish productivity in bodies of water. By accumulating in the organism of commercially valuable hydrobionts, pollutants represent a serious threat to human health. Their accumulation in the higher links of trophic chains leads to a subsequent reduction, a gradual or irregular decline in the latter and, consequently, to a restructuring of biocoenoses leading to a predominance of small hydrobionts not included in [these?] trophic chains.

In recent decades the issue of protecting the natural resources of northern seas has been widely discussed at various levels. Huge sums are being spent on development of research in this sphere. A great deal of experience has already been accumulated in studying the behavior and toxicology of pollutants.

Countries with highly developed industry have adopted strict laws regarding protection of the atmosphere, hydrosphere and lithosphere against poisonous wastes. But many unanswered questions remain.

We continue to experience the consequences of industrial pollution of those 111/ years when no appropriate measures had been undertaken to protect the environment. In addition, the rapid development of the chemical industry, application of chemicals in agriculture, and development of oil production on the shelf of northern seas are contributing to a situation in which new toxic compounds are being released into the aquatic environment every year. And although the threat of pollution in this region is being greatly reduced by improvements in the way pesticides are utilized, by restrictions adopted in a number of Northern European countries on the use of especially toxic persistent substances, and by enactment of legislative measures to protect waters against oil pollution, local anomalies in any portion of the world's oceans influence the condition of neighboring regions and take on a global character due to the large-scale circulation and integral nature of the ocean's biological structure.

In order to obtain an optimum system of evaluation and forecasting for the condition of northern seas, scientists of all Arctic states must further develop biogeochemical and ecotoxicological research, the most important areas of which are:

research on the role of atmospheric transport of contaminants;

study of the ways in which microimpurities are distributed and concentrated in marine ecosystems;

experimental study of the effect of pollutants on marine organisms of various taxonomic groups;

study of the ecological consequences of pollution;

study of the biogeochemical cycles of contaminants in marine ecosystems;

study of natural processes of self-cleaning and xenobiotic detoxification mechanisms;

formulation of principles and methods of biological indication for low levels of chronic pollution, and setting of standards for allowable concentrations of toxicants in the marine environment.

Fulfilling such a program of research will provide a scientific foundation for solving practical issues involved in protecting the purity of northern seas.

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[24] The Russian 'chastykovye' refers to fine-mesh nets and, by extension, to the fish for which these nets are used (Tr.).

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[25] This could be read either as the Arabic numeral '11' or as the Roman numeral 'II' (Tr.).

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